PROCEEDINGS OF THE ROYAL SOCIETY OF QUEENSLAND

BUSHFIRE 2006 CONFERENCE

SPECIAL EDITION

APRIL 2009 VOLUME 115

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We wish to acknowledge the assistance of the many professionals who gave their time as anonymous referees.

APRIL 2009

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COVER ILLUSTRATION: Recent prescribed burning in pasture lands (Prats de Molló, Pirineus, Catalonia). Source: Department of the Interior, Generalitat de Catalunya.

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ISSN 0080-469X

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Typeset by Francis Wild
Printed by Watson Ferguson & Company
1/655 Toohey Road, Salisbury, Qld 4107

PROCEEDINGS OF THE ROYAL SOCIETY OF QUEENSLAND
BUSHFIRE 2006 SPECIAL EDITION

SPECIAL EDITION PREFACE

Bushfire 2006 (held in Brisbane between 6-9 June 2006) marked the 10th in the series of Australasian Bushfire Conferences and the first time this conference series has been held in Queensland.

The first Bushfire Conference was held at the Australian Defence Force Academy in Canberra in 1987. These conferences are usually held biennially, with the last conference in Adelaide in 2004. The purpose and intent of these conferences has certainly evolved as we have increased on our knowledge and understanding of fire and its role in the environment. Early Bushfire Conferences were based on fire behaviour and modeling. Bushfire-93 in Perth started to look at the role of fire in the landscape, an area which would be a strong focus for Bushfire Conferences to follow, including Bushfire2006. It would be safe to say that the Bushfire Conference series provides a forum to discuss current research, development, techniques and innovation in the field of fire ecology and management.

Bushfire2006 had the theme - “Life in a Fire-Prone Environment: Translating Science into Practice”. Our main aim was to provide a platform for current research to be translated into useable operational guidelines and recommendation for land managers to incorporate findings into their management plans. We had five broader sub-themes:

- Translating science into practice - ecological bushland management,
- Development planning and control in fire-prone regions,
- Managing fire in modified landscapes (dealing with fragmentation, the role of fire in maintaining/establishing remnant connectivity, balancing production and biodiversity outcomes,
- Facilitating increased community consciousness towards bushfires, and
- Managing bushfire in a changing climate

Over three days, we had a total of 64 presentations, 19 well-presented posters, and 3 plenary addresses from Profs. Rob Whelan (University of Wollongong), Ross Bradstock (NSW Dept. of Environment and Climate Change, now University of Wollongong) and Dr. A. Malcolm Gill (CSIRO Plant Industy and the Bushfire CRC).

The papers presented in this special edition faithfully represents the quality and variety of papers presented during the conference, many with a strong ecological focus but providing outcomes which are useful to all land managers. The original theme on modeling and behaviour is also strong in this conference, with a number of papers investigating new models to better explain, for example the growth of fires, providing information for suppression activities. We also have a number of planning-related papers, focused on natural resource management, human resources or spatial data which tries to investigate optimising fire management for the conservation of biodiversity within the imperatives of the protection of life and property. There are also a number of quite reflective and philosophical papers on the current state of knowledge of fire ecology in the landscape. In my opinion, these papers provide a useful crystal ball to look into the future for fire ecology research.

I must take this opportunity to offer my deepest thanks to the Conference Organising Committee for their tireless work leading up to the conference, to ICMS Pty Ltd for their professionalism in helping with the organisation, the SEQ Fire and Biodiversity Consortium for such enthusiastic encouragement and championing of the conference. Finally, I wish to thank Jan Gilroy, my Research Assistant and Cate Melzer, Secretary for the Royal Society of Queensland who have gone beyond the call of duty to see this Special Edition of the Proceedings of the Royal Society of Queensland into reality.

Cuong Tran
Griffith School of Environment, Centre for Innovative Conservation Strategies, Griffith University
Editor– Bushfire2006 Conference Special Edition of the Proceedings of the Royal Society of Queensland

GUEST EDITORIAL by A. MALCOLM GILL

BUSHFIRES AND LANDSCAPES: TRANSLATING SCIENCE INTO PRACTICE

The papers in this Special Issue have arisen from the Conference "Bushfire 2006" under the subtitle "Life in a Fire-prone Environment: Translating Science into Practice". This suitably broad topic allowed discussion of a range of topics such as fires at urban-rural interfaces, effects of fire on plants and animals and the effects of climate change on fires. How can the information and ideas presented in this conference or those already published be translated into practice?

'Practice' in this context has to do with fire-prone land managed for particular outcomes. Since European settlement in Australia the general trend in land use has been from the 'natural' condition to pastoralism or forestry, thence to more intensive uses such as farming, plantation products, viticulture and orchards. Even more intensive land uses than these may follow one or other of these stages – from housing or industrial uses to the building of cities. Recognising the land use – reflecting the assets at hand – is the essential first step in translating our science into management practice.

Science seeks to develop an understanding of the processes and components of natural and artificially-modified systems of the world and the universe. It works because it is continually updating and revising its information in an impartial way. Initial proposals about any one topic may be modified through time, sometimes drastically, on the way to developing a new consensus – which, itself, may change in time. Rapid adoption of scientific findings to land management may be premature, therefore, unless the findings have been tested in some way. Have these findings been published in the scientific literature? Has there been debate concerning them? Has there been independent scientific acceptance of them? If the situation warrants urgent attention and early adoption, then the methods of science can be applied by managers to test the applicability of the findings locally, perhaps in association with scientists.

To apply the findings of science implies that the land manager has the necessary data to allow their implementation. Without knowing the history of fires, a scientific finding dependent on the history of fires for its application cannot be applied, for example; this is simple logic. Taking this example further, it becomes important for the application of science to land-management practice that all concerned are aware of the importance of the data and why it is essential to collect it accurately and completely. Furthermore, awareness as to what would be value-added data – such as fire severity and fire intensity – would help.

There may be the will to apply the findings of science but not the resources necessary to apply them. Managers often lament shortages of time, staff and money. Perhaps scientists and managers can work together to seek the necessary funding to allow new discoveries to be acted upon in a timely way. This may be difficult as the constraints on the scientist are just as real as those on managers.

If agencies are to effectively manage their core business - as reflected in targeted objectives - core data needs to be gathered to support it, appropriate analyses need to be performed on them, outputs of analyses need to be rapidly disseminated to appropriate staff, policy implications need to be addressed and then appropriate changes made to management practices. In this way, managers gain more knowledge and conduct more effective practices – and, in doing so – can also contribute to better science.

Translating science into practice is one side of the quest for better management of our landscapes. The other side of the quest is the scientific analysis of management data in order to gain a greater understanding of the environment. 'Translation' is facilitated if scientists and managers are able to understand the frameworks in which each operate; better landscape management is the outcome.

A. Malcolm Gill
CSIRO Plant Industry, Canberra; Fenner School of Environment and Society, ANU, Canberra and Bushfire CRC, Melbourne

USING HEAT AND SMOKE TREATMENTS TO SIMULATE THE EFFECTS OF FIRE ON SOIL SEED BANKS IN FOUR AUSTRALIAN VEGETATION COMMUNITIES

MANDA J. PAGE

Page, M.J. 2008 06 25: Using Heat And Smoke Treatments To Simulate The Effects Of Fire On Soil Seed Banks In Four Australian Vegetation Communities. *Proceedings of the Royal Society of Queensland*, 115: 1-9. Brisbane. ISSN 0080-469X.

The aim of this project was to investigate the ability of artificially applied smoke and heat stimuli to simulate field responses of soil seed banks to fire. The ability to simulate field responses to fire would allow managers to better understand the effects that fire might have on a community before actually burning it. This knowledge is greatly lacking for many Australian ecosystems at present, such that fire is avoided as a management tool for fear of negative ecological impacts. The composition and density of seedlings was compared between soil samples subjected to fire in the field and soil samples treated with a range of heat and smoke stimuli in 4 different Australian ecosystems along a rainfall gradient from subtropical to arid. The treatments used were: no treatment; smoke for 1 hour; heat at 80ºC; heat at 80ºC and smoke for 1 hour; heat at 105ºC; heat at 105ºC and smoke for 1 hour. The results vary greatly between ecosystems with no one treatment closely simulating the effects of fire. In addition, there was no significant difference between the control and the fire treatment in terms of the number of species or the abundance of seedlings that emerged from the seed bank. Thus the role of fire and fire related cues in stimulating seed banks is discussed. The results also revealed a relationship between seed bank response and the likely "natural" fire frequency of an ecosystem which may be useful to identify optimal fire intervals when managing for biodiversity. However, this needs further investigation.

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Fire is an important evolutionary factor in many Australian ecosystems (Kemp, 1981; Singh *et al.*, 1981) however fire regimes have been drastically changed since European settlement with the near exclusion of fire in many places (Gill, 1981). There is increasing recognition that fire is a useful management tool to prevent wildfires, manage specific plants such as endangered species and weeds, and promote biodiversity (Bond & van Wilgen, 1996). The problem is that there is generally insufficient information to determine the most appropriate fire regimes (Tran & Wild, 2000; Gill *et al.*, 2002). In addition, experimenting with fire in situ is spatially and temporally consuming, and fire at inappropriate times, intensities and frequencies can have detrimental effects (Gill & Bradstock, 2003). For reasons such as this, some conservation organizations advocate a precautionary principle approach to fire management.

Understanding the effect fire has on the soil seed bank is as important as understanding the influence of fire on the standing vegetation (Hill & French, 2003). The soil seed bank provides a reserve of seeds capable of replenishing a community following the loss of mature plants (Thompson, 1978; Bell, 1999) and this is particularly valuable after a fire. Fire is vital in triggering and facilitating successful seed germination and seedling establishment (Whelan, 1995) and many Australian plant species have become so adapted to fire that they will only geminate after fire (Bell, 1999; Dixon *et al.*, 1995). In Australia seeds of many species have seed dormancy overcome by fire, with the two main stimuli shown in laboratory tests to be heat and smoke (Enright *et al.*, 1997; Read *et al.*, 2000; Hill & French, 2003).

The aim of this study is to determine the ability of artificially applied smoke and heat stimuli to simulate seed bank field responses to fire in a range of vegetation communities. The composition and density of seedlings which emerge from soil seed bank samples in a range of ecological communities after fire will be compared with glasshouse germination from soil samples treated with heat and smoke stimuli.

TABLE 1. Characteristics of the study sites

Site Name	Vegetation Description	Rainfall (mm/yr)	Fire History	Date Sampled	Date Burnt
Fraser	Open mixed eucalypt forest with a scribbly gum (*Eucalyptus racemosa*) and pink bloodwood (*Eucalyptus intermedia*) overstorey and dense mixed understorey	1400	Frequently burnt, approx. every 5 yrs	5/04/02	9/08/02
Pine Rivers	Open eucalypt / Casuarina forest with a sparse midstorey of grass trees (*Xanth-orrhoea* spp.) and a dense understorey of kangaroo grass (*Themeda triandra*) and *Lomandra* spp.	1110	Unknown	6/08/02	10/08/02
Roma	Open eucalypt / Callitris woodland with a patchy midstorey of grass trees (*Xanthorrhoea* spp.) and a sparse grassy understorey	599	Wildfire in 1995	14/09/02	14/09/02
Currawinya	Open mulga (*Acacia aneura*) woodland with a low shrub stratum dominated by *Eremophila* spp. and a sparse ground cover dominated by *Eragrostis* spp.	332	No evidence of fire, last fire at least 50 yrs ago	18/02/02	20/02/02

MATERIALS AND METHODS

STUDY SITES

This study was undertaken in 4 different Queensland environments; an open mixed eucalypt forest community on Fraser Island (25º 30' 38.38S 153º 07' 48.46E), an open eucalypt woodland in south east Queensland's Pine Rivers Shire (27º 18' 20.69S 152º 53' 29.97E), a shrubby open woodland in a proposed scientific area approximately 30km north of Roma (26º 14' 32.87S 148º 49' 59.55E), and a tall open mulga shrubland on Currawinya National Park in south west Queensland (28º 49' 95S 144º 29' 09E). The characteristics of each study site are presented in Table 1.

SAMPLING

In each of the four vegetation communities, two one hectare replicates were selected which were representative of the vegetation community, as homogeneous as possible, and at least 50 meters from roads, tracks, water points and other forms of disturbance. Ten plots were randomly selected in each replicate and a 1m radius circular quadrat was marked at each plot. The percentage cover of each species in each plot was recorded along with the percent cover of litter and bare ground. Six random soil samples (17cm x 9cm and 2.5cm deep) were collected from the eastern half of each plot. A soil sampling tool was designed and used to extract a block of soil that remained in tact as much as possible. The reason extracting an in tact block rather than taking smaller, bulked samples was because the aims of the research were to attempt to simulate the effects of fire on the soil seed bank Each sample was placed in an aluminium tray that was filled within 3cm of the trays lip with sterile sand (for drainage purposes). Trays were labelled and transported back to The University of Queensland, Gatton. This sampling technique allowed the seed bank samples to remain in situ rather than the more commonly used method of bulking the samples together in an attempt to simulate the actual conditions of the soil seed bank when it is burnt. This method has rarely been used. Only the top 2.5cm of topsoil was sampled as previous studies have shown that most of a seed bank is concentrated at the surface (Read *et al.,* 2000). Following the fire, an additional sample was taken from the western side of each plot.

TREATMENTS

The soil samples from each plot were randomly allocated a treatment;

- Control – no treatment.
- Smoke – smoke aerially applied for 1hr.
- H1 – heated to 80ºC for 1hr.
- H1+S - heated to 80ºC for 1hr. then smoke aerially applied for 1hr.
- H2 – heated to 105ºC for 1hr.
- H2+S – heated to 105ºC for 1hr then smoke aerially applied for 1hr.

The sample collected after the fire was the "Fire" treatment.

The heat treatments were applied by placing the samples in a preheated oven for 60 minutes. This

method of 'dry' heat was used rather than 'wet' heat as it is more like the actual conditions of a fire (Enright *et al.*, 1997). The temperatures were chosen because heating at 80°C for 1hr resulted in the top 1cm of soil reaching an average temperature of 60°C, while the 105°C heating allowed the soil to reach on average 80°C. These are considered to represent low and medium to high intensity fires respectively (Humphreys and Craig, 1981). However, the extent to which seeds are heated within the soil seed bank depends upon their position within the soil, fire intensity and soil moisture (Enright *et al.*, 1997; Auld *et al.*, 2000).

The smoke treatment was applied aerially using a similar method to that of Dixon *et al.* (1995) and Read *et al.* (2000). The aerial application was used as it is a more effective form of smoke for germination than other methods such as smoke water (Lloyd *et al.*, 2000). The samples were placed in a plastic tent into which cooled smoke was forced for 60 minutes. The smoke was produced in a drum from the slow combustion of dry and green foliage, cooled in a long pipe wrapped in wet hessian and forced into the tent. The dry foliage was leaf litter collected from the sites, while the green foliage was eucalypt and acacia branches collected locally.

The prescribed fires were undertaken as part of the normal management practice and in accordance with fire management policies and plans for each location. All fires were lit during cool periods and were considered by management as low intensity, fuel reduction burns. The only diction where fire would not normally have been applied was the Currawinya site where prescribed fire is not applied as a management tool. In this situation, fire was applied during summer when there was deemed enough fuel and the best conditions to carry the fire (Table 1).

ASSESSING SEED BANKS

All soil samples were placed in the glasshouse directly after treatment. Positions were randomly rearranged twice weekly to decrease any effect of variation in the glasshouse. The samples were watered twice a day for five minutes from an overhead irrigation system. Each seedling that emerged was recorded, identified to species level (where possible) and removed. If seedlings could not be identified, they were repotted and grown until identification was possible. The samples were observed for 12 weeks but few emerged after 10 weeks.

DATA ANALYSIS

In three of the sites the seedling abundance and species richness data required transformation to achieve normal distribution. The Currawinya site data was normally distributed, the Roma site data required log transformation and the Pine Rivers and Fraser sites required an arcsinh transformation. The seedling abundance and total number of species in each of the treatments was compared for the two replicates separately as well as for the entire site using one-way analysis of variance (ANOVA) with Tukey's honestly significant difference (HSD) post hoc comparison of means tests. Two-way ANOVA's were also used to test the independent influence of the artificially applied smoke and heat on the treatments. When a significant difference is reported, $P<0.05$.

RESULTS

Overall 9068 seedlings from 125 species emerged during the study. The number of seedlings varied greatly between vegetation communities ranging from 5063 in the Roma site to 254 in the Fraser site (Fig. 1a). There was less variation in the number of species between vegetation communities with a range between 24 and 37 (Fig. 1b).

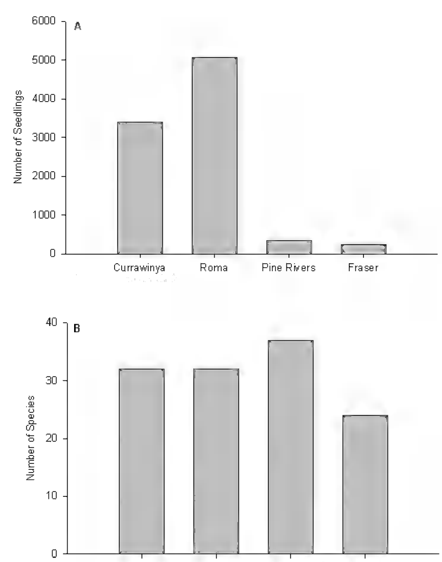

FIG. 1. Total number of seedlings (A) and species (B) recorded in each site

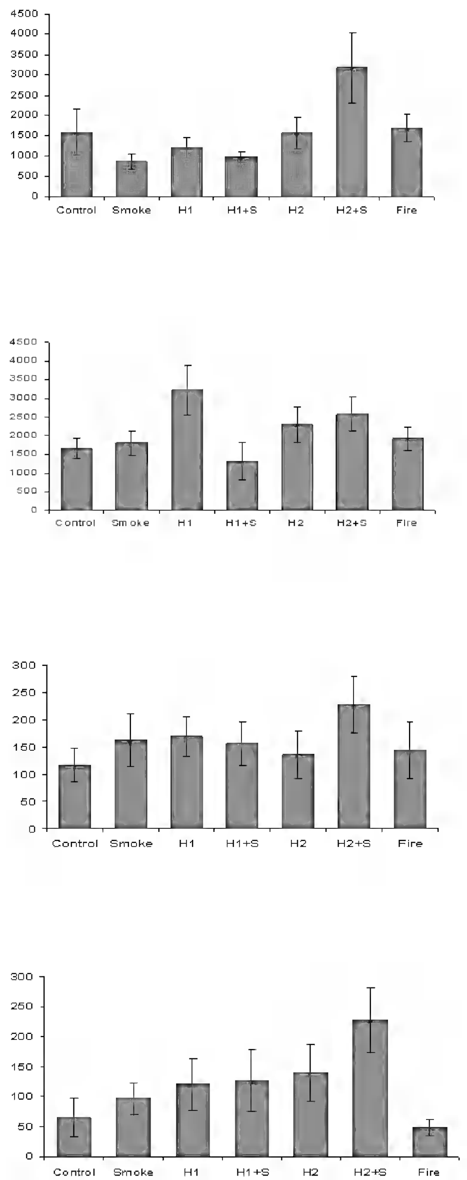

FIG. 2. Mean number of seedlings (± SE) per square meter for each treatment in A) Currawinya, B) Roma, C) Pine Rivers and D) Fraser communities. (Note: different Y axis scale between A&B, and C&D)

The response to treatments varied between the different communities. The mean number of seedlings per m2 that emerged for each treatment in each of the communities is presented in Figure 2. There was no significant difference in the number of seedlings between treatments in the Roma or the Pine Rivers sites. There was a significant difference detected in the Fraser site between the Fire and the H2+S treatments only. In the Currawinya site there was a significant difference between the H2+S and two other treatments; Smoke and H1+S. The H2+S treatment yielded the most seedlings in all communities except the Roma site where it was second greatest.

The number of species recorded for each treatment in each site is presented in Figure 3. There was no significant difference in the number of species between treatments in the Currawinya, Roma or Pine Rivers sites. However, a significant difference was detected in the Fraser site between the H2+S treatment and all other treatments except both H2 and Smoke (Fig. 3). The H2+S treatment yielded the greatest number of species in the Fraser and Pine Rivers sites. Currawinya was the only site where the Control yielded more species than any of the treatments and the only site which had less species in the Fire treatment compared to the Control. But, the Fire treatment did not yield the most species in any of the communities. The proportion of species found in the Fire treatment but not in the Control was greatest in the Pine Rivers site (40%), then the Fraser site (36%), the Roma site (33%) and lastly the Currawinya site (12%).

There was a significant difference in both the number of seedlings and the number of species between the

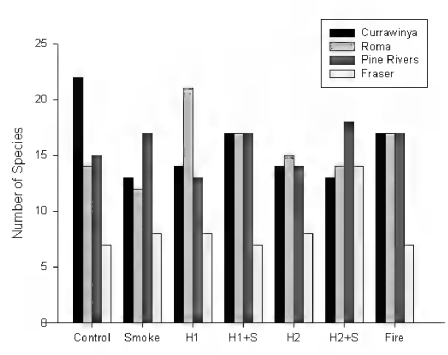

FIG. 3. Total number of species for each treatment in each site.

replicates in the Fraser community only. However, there was no significant difference between replicates within treatments in any of the communities based on either the number of seedlings or the number of species.

The species composition of the seedlings which germinated from the seed banks were dominated by forb species in each vegetation community, followed by grass species (Table 2). This trend was repeated in the individual treatments, with forbs dominating each treatments in each community. The proportion of species that were found in the control but not in the fire treatment varied between vegetation communities but ranged from 12 to 40% (Table 2).

DISCUSSION

The seed densities in this study ranged from 49 to 3172 seeds per square meter which is within the range of seed bank densities documented in other Queensland studies (Table 3). The lowest densities were found on the Fraser site and the highest densities were found in the semi-arid sites. This is consistent with the literature with the highest reported seed densities in Queensland being from the semi-arid south-central region (Navie *et al.,* 1996) and very low densities being reported in coastal woodlands (Drake, 1979). The seed banks investigated in this study were dominated by forbs and grasses with low numbers of trees and shrubs, again consistent with other Queensland studies (e.g. Navie *et al.*, 1996; McIvor *et al.*, 2004; Williams *et al.*, 2005). The lack of trees and shrubs is likely due to the longevity of these species which provides for population persistence, rather than investing in seed bank storage (Bond & Midgley, 2001).

It has been reported there is an obvious role for a regime of fire in nearly all Australian vegetation communities (Good, 1981; Cary *et al.*, 2003) and there is mounting evidence of the high number of native species whose dormancy is broken, or germination improved, by the properties of fire (Dixon *et al.*, 1995; Lloyd *et al.*, 2000; Read *et al.*, 2000). Willams *et al.* (2005) found that fire plays a crucial role in breaking seed dormancy in a range of Queensland savanna species. However, this study revealed that there was no significant difference between the number of seedlings, or the number of species, that emerged from the soil seed bank after fire compared to the Control in any of the vegetation communities investigated. Though not significant, there was a pattern of more species in the Fire treatments compared to the Control in each vegetation community except Currawinya. Furthermore, there was a difference in the composition of the species that emerged from the Fire treatment in comparison to the Control with a range from 12-40% of the species found in the Fire treatment not found in the Control.

The treatment with the highest temperature coupled with smoke (H2+S) yielded the highest number of seedlings in all communities (except Roma where it

TABLE 2. Number of species in each growth form group and the proportion of species identified in the Fire treatment but not the Control treatment for each vegetation community.

Site Name	Forb	Grass	Shrub	Tree	Fire but not Control Treatment
Fraser	12	8	3	1	36%
Pine Rivers	16	14	4	3	40%
Roma	17	11	2	2	33%
Currawinya	17	9	4	2	12%

TABLE 3. Germinable soil seed bank densities (per square meter) from Queensland studies.

Reference	Location of Study	Community Type	Seed Density Range (m^2)
Drake (1979)	South-east Queensland	Coastal Eucalypt forests	28-433
Hopkins *et al.* (1990)	North Queensland	Rainforest	434-4758
Clifford *et al.* (1995)	Southern Queensland	Eucalypt forest (from coast to semi-arid)	8-7432
Grundy (1996)	South-east Queensland	Eucalypt forest	231-7021
Navie *et al.* (1995)	South-central Queensland	Subtropical Eucalypt woodland	9907-17261
Odgers (1996)	South-east Queensland	Eucalypt forest	114-1745
McIvor *et al.* (2004)	South-east Queensland	Native pastures	6000-12000
Williams *et al.* (2005)	North-east Queensland	Tropical Eucalypt forest	50-792

was the second highest) and was present each time a significant difference was detected between treatments. Williams *et al.* (2005) also found that heat and smoke treatments significantly increased the germination of several species. But, this treatment yielded more seedlings than the Fire treatment in this study. This suggests that the prescribed fires applied in this study did not provide their full potential of heat and smoke stimulation on the soil seed bank. The burns could have been too low in intensity or too patchy to affect the seed bank evenly. Unfortunately these variables were not measured but all of the burns conducted for this study were low in intensity for either safety reasons or because fuel loads were insufficient.

Apart from the H2+S treatment consistently yielding the highest results, there were no other consistent patterns between communities in relation to the response to the different treatments. No studies were detected which compared results between different ecosystems and thus it is unclear whether this result is unusual. This may be because fire plays a different role in each of the communities investigated. For example, the fire treatment was most similar to the H2+S treatment in the Currawinya site and the Smoke treatment in the Roma site, two very different treatments. The composition of the species within each community was very different and thus may respond to, or be triggered by, different aspects of fire. Alternatively, fire may not be as important in these ecosystems as was initially assumed. Given there was no significant difference in either the number of seedlings or the number of species detected between the Control and the Fire treatments in any of the vegetation communities indicates that fire may not have a big impact on what emerges from the seed bank. Fire has many other effects on a community such as altering the structure of the vegetation, the composition and abundance of the standing vegetation, the soil chemical and physical properties, light availability and intensity, and removing allelopathic substances. We know that many Australian plants have adaptations that allow the standing vegetation to regenerate after fire and thus the impact on the soil seed bank may be insignificant in comparison. However, a number of unique species were detected with fire and this is important to consider.

The sampling design and effort used in this study should also be considered when discussing results as soil seed banks are highly variable both temporally and spatially (Fenner, 2000) and often high numbers of samples are required to accurately represent the soil seed bank (Page *et al.*, 2006). Using the post hoc method suggested by Page *et al.* (2006) to evaluate the adequacy of soil seed bank studies, on average the number of species undetected was around 30%. This is a relatively high proportion of undetected species and may indicate that sampling effort per treatment was insufficient to fully detect patterns. Furthermore, the germination technique used here has some limitations as not all species in the seed bank can be detected, only those whose germination requirements are met by the treatments. However, this method has been shown to be a reliable estimate of the viable seeds in a sample (Gross, 1990).

A pattern was revealed which related to the longitudinal location of the sites studied. The number of seeds was greater in the more western sites (Roma and Currawinya) than the eastern sites, and the proportion of species found in the Fire treatment increased as the location moved east. The annual average rainfall and its predictability also increases as the site location moves east. As the amount and predictability of rainfall changes so too does the type of vegetation, and thus the likely fire regimes. That is, in arid systems fire is likely to be infrequent and unpredictable while in higher rainfall areas fire is likely to be more frequent as biomass is accumulated more quickly, and more predictable with distinct wet/dry seasons. Fenner and Thompson (2005) state that seed banks are most advantageous in communities with frequent catastrophic disturbances that are unpredictable, and that the severity and predictability of the disturbance determines the persistence of a seed bank. Fire may be identified as such a disturbance and thus the seed bank should increase as fire (the disturbance) becomes more frequent. But as fire becomes more frequent, it also becomes more predictable. In these ecosystems it seems that the more predictable and frequent the fire regime is likely to be, the less reliant the community is on seed bank. It should also be noted that fire is not the only severe disturbance that impacted on these ecosystems, the most obvious being drought in the arid and semi arid areas. So although the disturbance of fire is infrequent and unpredictable, the disturbance of drought is frequent, though still somewhat unpredictable. Considering this, Fenner and Thompson's (2005) prediction of seed banks being more important in communities that have frequent but unpredictable, severe disturbance holds true with Currawinya and Roma having the highest seed bank records.

Although the number of seeds was higher in the sites with less frequent and unpredictable fire, the number of species that were found only in the Fire treatment increased as fire frequency increased. So although areas with infrequent and unpredictable disturbances rely heavily on seed banks and thus had a much greater seed bank, the areas with more frequent and predictable fires had more species in the seed bank that were reliant on fire for germination. This is logical as why have seeds in the seed bank that need fire to trigger or enhance germination when fire is unlikely to be a frequent or predictable event? However, most plants are triggered to germinate by rainfall in arid systems (Inouye, 1991) but rainfall is unpredictable and infrequent in these ecosystems so why have species triggered by rainfall when it is infrequent and unpredictable? The difference is that moisture is a necessary resource for almost any plant to germinate where fire may be a factor that simply allows a competitive advantage. In the moister communities the canopy is denser and germination may be more reliant on a gap in the canopy than a rainfall event and fire can create such gaps. Thus having germination cues relating to fire may be extremely beneficial as it is more likely that gaps will be present post-fire. Currawinya is the most arid site investigated in this study and it was the only site which had the highest number of species in the Control. This is a further indication that this is not a fire triggered system. Similarly Thomas *et al.* (2003), found that the germination of species from habitats that are infrequently burnt are not affected by heat shock or smoke.

Another explanation of this longitudinal pattern in the size of the seed bank is that fire depletes the seed bank and thus areas with more frequent fire are unlikely to have abundant seed banks. From our study sites, fire is the most frequent on Fraser Island with a fire occurring on average every 5 years. This could explain the low seed bank encountered on Fraser sites.

Testing hypothesis about the relationship between fire frequency, predictability and soil seed banks were not the aim of this study and thus are only speculative. Further studies which specifically aim to test these hypotheses are required.

ACKNOWLEDGEMENTS

Thanks to Alicia Whittington and Janet Newell for undertaking much of the field work, and Danny McKellar, Kristy Lawrie and Craig Welden for arranging burns. Sean Bellairs was instrumental in developing the proposal and gaining funding for the project through The University of Queensland Research Development Grant Scheme.

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CONCEPTS, CHARACTERISTICS, COMPETITION: TOOLS IN THE SEARCH FOR SUSTAINABLE FIRE REGIMES

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Watson P. 2008 06 25: Concepts, Characteristics, Competition: Tools In The Search For Sustainable Fire Regimes. *Proceedings of the Royal Society of Queensland*, 115: 11- 21. Brisbane. ISSN 0080-469X.

Fire ecology researchers risk bombarding managers with a 'blizzard of detail'. Over the past decade, there have been various efforts to synthesize information from ecological research into management guidelines; in particular, ecologists have sought to provide guidance as to fire frequencies compatible with retention of species in different vegetation types. This paper takes a step back, and uses some of the ecological concepts and models which underpin understanding of the role of fire in fire-prone ecosystems to explore issues around appropriate fire frequencies for biodiversity conservation. Concepts include disturbance, succession, plant species vital attributes, interspecific competition, landscape productivity and patch dynamics. Ideas are illustrated through discussion of the dynamics of a number of vegetation types including a subcoastal grassy woodland, a wet sclerophyll forest, and a coastal heath.

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The by-line for the Bushfire 2006 conference in Brisbane, for which this paper was developed, was "translating science into practice." However the science of fire ecology advances not through Noble-prize-winning leaps, but incrementally through many painstaking studies that range across a multitude of species, ecosystems, and aspects of fire. How can this wealth of information be transmuted from a confusing 'blizzard of detail' into a form which managers can use?

I believe there is value in explicitly presenting and discussing ecological concepts and models which underpin scientific understanding of the dynamics of fire-prone ecosystems. This exercise can assist scientists to produce sound management recommendations; help scientists, managers and advocates to understand the considerable differences between vegetation types and regions; and assist stakeholders to find common ground where conflict currently exists. Of course, concepts and models are "abstractions of reality, caricatures" (Gill & Bradstock, 1995:311), and their ability to describe what happens in the real world is necessarily limited. Still, they provide frameworks for understanding, and thus have the potential to contribute considerably to the search for sustainable fire regimes.

In this paper I use a series of ecological concepts to explore issues around the question of appropriate fire frequencies for biodiversity conservation. These thoughts represent my understanding after years of reading, discussing, researching and writing about fire frequency in shrubby and grassy ecosystems in south-east Queensland and New South Wales (NSW) (Watson, 2001, 2005, 2006; Watson & Wardell-Johnson, 2004). My primary aim is to stimulate discussion and research; these notions are not fully formed and of course are amenable to change and development. The ideas presented here draw on recent work in the NSW Department of Environment and Conservation (DEC) on the development of fire regime guidelines (Bradstock & Kenny, 2003; Kenny *et al.*, 2004), and a secondary aim of the current paper is to suggest directions in which this work might usefully be extended.

Ecological concepts and principles which underlie current understanding of fire regimes include disturbance, succession, plant species vital attributes, interspecific competition, landscape productivity and patch dynamics.

DISTURBANCE, SUCCESSION, AND A PARADIGM SHIFT

Fire is a disturbance. A disturbance can be defined as "any relatively discrete event in time that removes

organisms and opens up space which can be colonised by individuals of the same or different species" (Begon *et al.*, 1990). The concept encompasses recurring discrete events such as storms, floods and fires, as well as on-going processes like grazing. Disturbance may stem from natural phenomena or human activities (Hobbs & Huenneke, 1992), and is ubiquitous throughout the world's ecosystems (Sousa, 1984).

When disturbance is substantial, succession follows. The concept of succession has been of interest to ecologists since Clements outlined what is now called 'classical succession' in 1916. In classical succession "following a disturbance, several assemblages of species progressively occupy a site, each giving way to its successor until a community finally develops which is able to reproduce itself indefinitely" (Noble & Slatyer, 1980:5). Implicit in this model is the idea that only the final, 'climax' community is in equilibrium with the prevailing environment.

A popular metaphor for this equilibrium paradigm is 'the balance of nature'. Conservation practice aligned with this model focuses on objects (e.g. specific species, 'vegetation') rather than processes, concentrates on removing the natural world from human influence, and believes that desirable features will be maintained if nature is left to take its course (Pickett *et al.*, 1992). Fire does not sit easily in the balance of nature approach, which influenced attitudes to burning, both in Australia and elsewhere, for many years. For example, forester C.E. Lane-Poole argued to the Royal Commission following the 1939 fires in Victoria for total fire exclusion on the grounds that this would enable natural succession to proceed resulting in a less flammable forest (Griffiths, 2002).

Over recent decades, however, a paradigm shift has been underway. Drivers include the realisation that multiple states are possible within the one community (Westoby *et al.*, 1989), as are multiple successional pathways (Connell & Slatyer, 1977; Turner *et al.*, 1993). Most importantly from a conservation perspective, it has increasingly been recognised that periodic disturbance is often essential to maintain diversity, allowing species which might otherwise have been displaced to continue to occur in a community (Connell, 1978).

The non-equilibrium paradigm can be encapsulated by the phrase 'the flux of nature'. Scale is important in this paradigm: equilibrium at a landscape scale may be the product of a distribution of states or patches in flux (Turner *et al.,* 1993; Wu & Loucks, 1995). Stringham *et al.* (2003:109) define a state as "a recognizable, resistant and resilient complex" which includes both soil and vegetation; in this paper the term is used to describe alternative vegetation communities that can occur on the one piece of ground.) Implications of the non-equilibrium paradigm include a legitimate – or even vital – role for people in ecosystem management, and a focus on the conservation of processes rather than objects. This does not, of course, imply that all human-generated change is acceptable; it does mean human beings must take responsibility for maintaining the integrity of ecosystem processes (Pickett *et al.*, 1992). Fire fits much more comfortably into the non-equilibrium paradigm, where it takes its place as a process integral to many of the world's ecosystems.

The concepts of disturbance and succession are basic to the science of ecology; they could even be considered too basic to rate a mention. However shifts in thinking over recent decades have not always made their way into the public arena. Adherence to the idea that fire is an undesirable impediment to the natural process of succession may underlie the attitudes of some people, including some sincere conservationists, to fire.

THEORY INTO THRESHOLDS

The non-equilibrium paradigm forms the basis for a number of theories and models which have been used to inform an understanding of fire regimes in Australia. These include the vital attributes model of Noble & Slatyer (1980). This scheme employs a small number of life history characteristics of plant species, termed 'vital attributes' because they are "vital to the role of the species in a vegetation replacement sequence" to predict successional pathways (Noble and Slatyer, 1980:6). It can also be used to define disturbance frequency domains compatible with maintenance of particular suites of species. The vital attributes model has recently been used to develop fire management guidelines for broad vegetation types in NSW (Kenny *et al.*, 2004), and is also used in Victoria (Friend *et al.*, 2003). Here is one way to present at least some of the 'blizzard of detail' in a manager-friendly form.

The basic idea is that, to keep all species in a community, fire intervals should vary within an upper and a lower threshold. Lower thresholds, which I will

call 'minimum interfire intervals for management', are set to allow all species vulnerable to frequent fire to reach reproductive maturity, while upper thresholds ('maximum interfire intervals for management') are determined by the longevity of species vulnerable to lack of burning. Species with similar fire-related attributes are grouped into functional types (Noble & Slatyer, 1980; Keith *et al.*, 2002b). The vulnerability of each group, and of species within sensitive groups, can be assessed through consideration of species' vital attributes.

Functional types most sensitive to short interfire intervals (high fire frequency) contain obligate seeder species whose seed reserves are exhausted by disturbance (obligate seeder species are killed by fire and rely on regeneration from seed). Populations of these species are liable to local extinction if the interval between fires is shorter than their primary juvenile period (Noble & Slatyer, 1980), the time from seed germination to reproductively mature adult. The minimum interfire interval to retain all species in a particular vegetation type (lower threshold) therefore needs to accommodate the taxon in this category with the longest juvenile period (DEC, 2002).

Species whose establishment is keyed to fire (Noble & Slatyer call these 'I species') are highly sensitive to long interfire intervals (infrequent fire): they are liable to local extinction if fire does not occur within the lifespan of established plants and/or seedbanks (Noble & Slatyer, 1980). The maximum interfire interval for management (upper threshold) therefore needs to accommodate the taxon in this category with the shortest lifespan, seedbank included (DEC, 2002; Bradstock & Kenny, 2003).

Data on plant vital attributes relevant to setting minimum interfire intervals for management – regeneration modes and juvenile periods – are reasonably easy to obtain. Predictions of shrub abundance based on these vital attributes have recently been tested through a field study of woodland vegetation in areas with known fire histories, and were supported (Watson, 2005). It should be noted that when thresholds are determined for a very broad geographic area, as is the case in the DEC guidelines for the state of NSW (Kenny *et al.*, 2004), minimum interfire intervals for management may be set by species which are not found in particular local areas, and thus may be above what they would have been had only local species been considered.

Data relevant to setting maximum interfire intervals for management (upper thresholds) – longevity of adults and seeds – are much less readily available. Kenny *et al.* (2004) note the lack of quantitative data on these attributes, and point out that as a result, maximum interfire intervals for management in the NSW guidelines are "largely based on assumptions and generalisations" and are therefore surrounded by "considerable uncertainty" (Kenny *et al.*, 2004:31). Work on these variables is an important task for the future. Additionally, in determining maximum interfire intervals for management for NSW vegetation types, "any data considered dubious was excluded from guideline calculations" (Kenny *et al.*, 2004:25). This included estimates in forms such as "life span possibly 20-30 years." Where a range of figures for lifespan was available, the highest figure was used (Kenny *et al.*, 2004:26). Decision rules have therefore tended to set maximum interfire intervals for management higher, rather than lower. It could be argued that application of the precautionary principle in relation to species liable to local extinction through lack of fire would imply that decision rules should reduce, rather than raise, thresholds at this end of the interval domain.

It can also be argued that maximum interfire intervals for management might be better set some years below lifespan + seedbank longevity (y) of species liable to be lost from a community if fire is insufficiently frequent. West Australians Burrows & Abbott (2003:446) suggest 0.75y may be more appropriate. Population decline, both above and below ground, may occur over a long period prior to the point of local extinction (Auld, 1987). Flowering may peak in the years following the juvenile period: McFarland (1990) found flowering and seeding in south-east Queensland's wallum heath peaked at four to eight years after a burn, and dropped markedly by 11 years post-fire. A species may therefore still occur in the landscape, but its fecundity might be greatly reduced in later post-fire years (Auld & Myerscough, 1986).

Finally, the potential for one or a small number of species to dominate under extended interfire intervals, and related competitive interactions, may mean that maximum interfire intervals for management need to be some years below what an assessment based solely on the vital attributes of individual plant species would suggest. This issue is explored in the next section.

COMPETITION AND PRODUCTIVITY

The effect of dominant heathland shrubs, such as *Banksia ericifolia* and *Allocasuarina distyla*, on other species has been recognised in Sydney's sandstone country (Keith & Bradstock, 1994; Tozer & Bradstock, 2002). When plant species vital attributes alone are considered, a feasible fire frequency for the conservation of both these dominant obligate seeders and understorey species appears to be 15-30 years. However at this fire frequency, the dominant species form high-density thickets which reduce the survival and fecundity of species in the understorey, an effect which carries through to the next post-fire generation (Keith *et al.*, 2002a, 2002b). Similar dynamics have been observed in other Australian heath communities (Specht & Specht, 1989; Bond & Ladd, 2001) and in South Africa's heathy fynbos (Bond, 1980; Cowling & Gxaba, 1990; Vlok & Yeaton, 2000). An understanding of this dynamic has highlighted the need to include in heathland fire regimes some intervals only slightly above the juvenile period of the dominant species, thus reducing overstorey density for a period sufficient to allow understorey species to build up population numbers before again being overshadowed (Keith *et al.*, 2002b). This need is reflected in fire frequency recommendations in Bradstock *et al.* (1995).

The competitive effect on understorey vegetation may be particularly profound where dominant shrubs resprout (Bond & Ladd, 2001). Unlike obligate seeders, dominant resprouters will continue to exert competitive pressure immediately after a fire by drawing on resources in the soil, and once their cover is re-established, on light resources too. These dynamics have been documented in the grassy woodlands of Western Sydney's Cumberland Plain, where the prickly shrub *Bursaria spinosa* (known to landholders simply as Bursaria) forms dense thickets which can dominate the landscape where fire has been excluded for several decades (Watson, 2005). After a burn, this shrub resprouts and grows rapidly. Other shrub species in this vegetation type, particularly obligate seeders, are less abundant in Bursaria-dominated landscapes than in sites which have burnt once or twice a decade, an outcome which probably reflects competitive pressure from Bursaria. Ground layer species are also affected. Thus the strategy recommended to provide relief for competitively inferior species in heaths – inserting one short interval amongst longer ones (Keith *et al.*, 2002b) – is unlikely to work in this community. Maximum interfire intervals for management need to be sufficiently low to allow for the moderately frequent fires that will let Bursaria thickets, obligate seeder shrubs and a diverse ground layer of native grasses and herbs co-exist long-term.

In Cumberland Plain Woodland, Bursaria – and some introduced shrubs such as African Olive (*Olea europaea* subsp. *cuspidata*; von Richter *et al.*, 2005) – have the unusual advantage of being able to recruit between fires, whereas most sclerophyllous (hard-leaved) shrub species recruit almost exclusively after a fire (Purdie & Slatyer, 1976; Cowling *et al.*, 1990; Keith *et al.*, 2002a). The vital attributes model explicitly identifies species able to recruit between fires – Noble & Slatyer call them 'T species'– and their propensity to dominate in the absence of disturbance is also explicitly noted (Noble & Slatyer, 1980). However to date little emphasis has been placed on the role of T species when determining fire frequency guidelines.

Currently, the vital attributes model is concerned with species presence or absence, not with issues of abundance or dominance. Noble & Slatyer (1980:19) recognise this limitation, and suggest that "to account for relative density… and other interactions, a more quantitative description of one or more vital attributes may be required." For example species could theoretically be ranked, Noble & Slatyer (1980) point out, in terms of their ability to recruit in the presence of established adults. A fourth vital attribute related to growth rate and size at maturity could be added.

The importance of competition between plant species, and thus the importance of disturbance to disrupt competitive exclusion, is likely to vary with landscape productivity. A second non-equilibrium paradigm offshoot, the dynamic equilibrium model, considers the interaction of productivity and disturbance in mediating species diversity (Huston, 1979, 2003, 2004; Kondoh, 2001). In harsh environments where productivity is low, interspecific competition is unlikely to be great. Here, a-biotic factors such as low rainfall, heavy frosts and infertile soils limit the number of plant species able to grow, and also limit their growth rates. The need for disturbance to reduce competitive superiority is therefore minimal. In fact, a high disturbance frequency is predicted to reduce diversity in these ecosystems, as organisms will be unable to grow fast enough to recover between disturbances. In highly productive, resource-rich environments, however, competition is likely to be

much more intense, as many species can grow in these areas, and they grow quickly. Here, diversity is predicted to decline where disturbance frequency is low, as some species will outcompete others, displacing them from the community.

Landscape productivity, as reflected in plant growth rates, is likely to increase with moisture availability − which is influenced by rainfall, temperature, and season of rainfall — and with soil fertility (clay soils are often more fertile than sandy soils, however they also tend to support more herbaceous, and fewer shrub, species; Specht, 1970; Prober, 1996; Clarke, 2003). If Huston's theory holds, relatively frequent fire may thus be more appropriate in wet, warm, productive fire-prone systems than in those whose productivity is limited by poor soils, low rainfall or a short growing season. This is because competitive exclusion may cut in more rapidly in warm, wet environments where growth rates are higher, increasing the need for periodic disturbance to open up space for non-dominant species.

The above reasoning may apply in both grassy and shrubby ecosystems, although the lifeforms of the plant species involved, and timeframes, will differ. Studies have shown the importance of periodic disturbance in the productive temperate grasslands of Victoria's basalt plains: here, the dominant grass *Themeda triandra* (Kangaroo Grass) forms a closed canopy by three years post-fire (Morgan, 1998), restricting space for the smaller herbaceous species which grow between its tussocks (Lunt & Morgan, 1999; Coates *et al.*, 2006). Interfire intervals as low as 1-3 years have been recommended to ensure gaps for interstitial species in this community (Morgan, 1998; Coates *et al.*, 2006). Competitive exclusion does not appear to happen so rapidly, however, in semi-arid grasslands, where lower moisture availability may slow tussock closure, indicating fires at such very short intervals are unlikely to be necessary to provide space for intertussock forbs and grasses (Lewis, 2006). In shrub-dominated communities canopy closure is likely to take longer than in grasslands subject to similar climatic conditions. Thus in relatively high rainfall coastal heaths around Sydney, it takes six or seven years for the dominant shrubs to overshadow smaller co-occuring species (Tozer & Bradstock, 2002). In less productive shrubby ecosystems competitive exclusion can be expected to take longer to manifest. For example in the mallee shrublands of the semi-arid zone, the dominant shrub *Callitris verrucosa* takes 10-15 years to mature, though by 100 years without a fire this species may exclude even mallee eucalypts, as well as smaller shrubs (Bradstock & Cohn, 2002). The dynamic equilibrium hypothesis would predict that interfire intervals compatible with a high diversity of shrubland plants are likely to be somewhat longer in the mallee than in more productive coastal heaths.

These examples suggest a second reason why appropriate interfire intervals for management may be shorter in more productive systems, and longer in less productive ones: plant species may reach life history milestones sooner where moisture availability and fertile soils allow for rapid growth. In particular, that key indicator of minimum interfire intervals for management, the juvenile period of obligate seeder shrubs, may be shorter where resources are more readily available. For example while *Banksia ericifolia* and *Allocasuarina distyla*, the dominant shrubs in Sydney coastal heath, mature by around seven years post-fire (Benson, 1985; DEC, 2002), the ecologically-equivalent species in the semi-arid landscapes of central NSW, *Callitris verrucosa,* has a juvenile period of 11-13 years (Bradstock & Cohn, 2002). Similarly, shrub juvenile periods on the New England Tablelands, where the growing season is constrained by severe frosts, can be several years longer than those of the same species in coastal areas (Knox & Clarke, 2004). A key indicator of maximum interfire intervals for management, time to senescence in plants with canopy-stored seed (Noble & Slatyer, 1980; Bradstock & Kenny, 2003; Burrows & Abbott, 2003), may also be shorter in more productive areas (Specht & Specht, 1989). Again comparing heathland and mallee: where the Sydney coastal heath dominants have lifespans of around 30-40 years (DEC, 2002; Bradstock & Kenny, 2003), the prominent mallee species *Callitris verrucosa* is believed to live "well in excess of 100 years" (Bradstock & Cohn, 2002).

A third reason why relatively short interfire intervals may be particularly important for maintaining complements of fire-adapted species in productive high rainfall areas, is because these environments may potentially host a relatively large pool of woody 'T species', adapted to recruitment in gaps rather than being cued to fire (I species). This is likely to be the case where landscapes contain a mosaic of fire-prone and rainforest vegetation. In wet sclerophyll forest on the New England Tablelands, Campbell & Clarke (2006) found that while most sclerophyll species

recruited from seed after fire, most broad-leaved taxa were gap rather than post-fire recruiters. Under long interfire intervals, rainforest species may use their ability to recruit between fires to move into adjacent sclerophyll forests and woodlands, outcompeting sclerophyll shrubs and light-loving grasses (Smith & Guyer, 1983; Bowman & Fensham, 1991; Woinarski *et al.,* 2004). When a fire does finally occur, these rainforest species may be remarkably persistent; studies in northern NSW and in Queensland have confirmed that at least in these environments, most resprout after fire (Williams, 2000; Campbell & Clarke, 2006). By contrast, dry sclerophyll forests in the less productive environments found on the NSW Southern Tablelands typically 'open out' some decades after fire as fire-cued shrubs senesce (Purdie, 1977). Here almost all shrub species recruit primarily after fire (Purdie & Slatyer, 1976), and competitive exclusion by large woody T species does not appear to be an issue (Watson, 2006).

This discussion brings us back to the concept of succession. South African fire ecologists Bond *et al.* (2003, 2005) divide global vegetation types into three categories. Climate-limited communities are not prone to either major structural change, nor to succeeding to another vegetation type in the absence of fire, although fire frequency may mediate species composition to some extent. In South Africa, these communities occur in arid environments, and also in areas nearer the coast where rainfall is moderate but occurs in winter. Climate-limited but fire modified vegetation types do not succeed to another vegetation type, in terms of the overall species composition and the dominant stratum, in the absence of fire, but their structure may alter from grassy to shrubby. The Cumberland Plain Woodland discussed above fits into this category. Fire-limited vegetation types will succeed to a different community in the absence of fire. In South Africa, these communities occur in higher rainfall areas, where a trend towards forest has been found in long-term fire exclusion experiments in both savanna and heath (Bond *et al.,* 2003).

Climate-limited but fire-modified systems can occur in at least two 'states', for example grassy woodland and Bursaria-dominated shrub thicket woodland on the Cumberland Plain (Watson, 2005). Fire-limited vegetation types could also be said to be able to exist in different states, although the differences between them (at their most extreme) are so great that they are rarely thought about in this way. For example, in north Queensland, *Eucalyptus grandis* grassy wet sclerophyll forest is succeeding to rainforest, probably due to a reduction in fire frequency and/or intensity (Unwin, 1989; Harrington & Sanderson, 1994). However rainforest and grassy wet forest are not generally considered as different states of a single vegetation type, but rather as two different types of vegetation.

PATCH DYNAMICS

Fire can mediate a landscape of different habitat patches, whose location may change over time. First, there is the mosaic reflecting variations in the time elapsed since the most recent fire – the time-since-fire mosaic. Then there is the mosaic reflecting the history of interfire intervals across the landscape – the fire-frequency mosaic. These two mosaics are related but not the same, and their consequences for vegetation composition and structure, and thus for fauna habitat, may be different (Morrison *et al.,* 1995; Watson & Wardell-Johnson, 2004; Bradstock *et al.,* 2005). Both mosaics are likely to play a role in maintaining diversity. In some landscapes, particularly the more productive ones whose composition and structure can vary widely, management aimed at keeping interfire intervals within a single set of fire frequency thresholds, that is variation in interfire intervals across time, may not be sufficient to maintain all ecosystem elements. Variation in fire frequency thresholds across space may also be needed to ensure all possible states, and the plants and animals they support, are able to persist in the landscape.

For example, recent studies in north-eastern NSW indicate that some forests in high rainfall areas on moderately fertile soils can exist in more than one 'state'. Relatively frequent fire – at intervals between 2 and 5 years – is associated with open landscapes in which a diverse flora of tussock grasses, forbs and some shrubs thrives (Stewart, 1999; Tasker, 2002). This landscape provides habitat for many invertebrate species (York, 1999; York, 2000; Andrew *et al.*, 2000; Bickel & Tasker 2004), and for a number of rare small mammals (Tasker & Dickman, 2004). Nearby areas which have remained unburnt for periods over 15 or 20 years support higher densities of some shrub and non-eucalypt tree species, particularly those able to recruit between fires (Birk & Bridges, 1989; York, 1999; Henderson & Keith, 2002). Litter levels are higher in these multi-layered areas (Birk & Bridges, 1989), which provide habitat for an equally diverse, but substantially different, array of invertebrates and

small mammals to that found in frequently burnt areas (Catling *et al.*, 2000; York, 2000; Tasker & Dickman, 2004).

Grassy wet sclerophyll forests in the sub-tropics can thus exist in at least two 'states.' The dynamic nature of these forests suggests they would fall into either Bond's 'climate limited but fire modified' or his 'fire limited' category (Bond *et al.*, 2003, 2005). Burning at short intervals limits the extent to which vegetation progresses down the path towards shrubbiness and high litter levels; whether successional change in the absence of fire would result in a transition to rainforest remains to be determined, and could well vary with rainfall, slope, aspect, geology and proximity to existing rainforest patches.

The existence of two understorey 'states' supporting diverse but distinct suites of species in the grassy wet forests of Northern NSW suggests the need for a fire regime which supports the continuation of each state somewhere in the landscape. In some places, fire needs to happen often enough to maintain open forest environments rich in grasses and herbs, where early-successional animal species can thrive. Other places need to support good-sized patches of thicker vegetation where mesophyll shrubs and late-successional fauna can flourish. For some fauna species, the juxtaposition of grassy and shrubby patches may be vital (Christensen, 1998). The vulnerable Palma Wallaby (*Macropus parma*) is an example (Maynes, 1977).

CONCLUSION

The concept of 'states' provides options for the creation and maintenance of habitat across space as well as time. It can reduce conflict between those who see the value in particular states (such as grassy or shrubby vegetation in sub-tropical wet sclerophyll forests), by pointing out the value of each and the need for both. Of course, it also raises questions as to the proportion of each state that may be desirable in the landscape, the scale of mosaics, the relationship between the time-since-fire and the fire frequency mosaics, the links between topography, landscape features and states, and the impact of fragmentation and climate change on these factors. These questions are beyond the scope of this paper, and some are probably beyond the scope of our current understanding; they represent fertile ground for research and discussion in the future.

Acceptance of the need for different fire frequency-mediated states in some vegetation types may also require a somewhat different approach to the determination and presentation of fire frequency thresholds (interfire intervals for management). Two or even three sets of thresholds may be needed to define frequencies predicted to characterise different states. It may be appropriate to consider the vital attribute-related requirements of some species and/or species groups, but not others, when proposing guidelines for particular states. The role of competition may deserve more explicit consideration in these labile, productive vegetation types than is necessary in 'climate limited' ecosystems.

The 'blizzard of detail' generated by the large number of studies addressing aspects of the relationship between fire, vegetation and fauna can be ordered to some extent when one recognises that fire regimes vary considerably between different vegetation formations (Bond, 1997; Watson, 2001; Kenny *et al.*, 2004). Concepts such as disturbance, succession, competition and productivity can help clarify why this is the case; they can assist both ecologists and on-ground managers to identify similarities and differences in the fire-related dynamics of different vegetation types, and to relate these differences to a-biotic factors. Ecological concepts can also inform the development and refinement of models linking characteristics of individual species to characteristics of ecosystems and disturbance regimes. Of course, models draw on empirical studies, and predictions from models, where possible, should be verified in the field (Cary *et al.*, 2003). Concepts, models, empirical research, historical and indigenous perspectives should all be grist for the mill in the important work of developing fire management guidelines.

ACKNOWLEDGMENTS

This paper has been informed by discussions of vegetation dynamics with many colleagues, particularly those associated with the South-east Queensland Fire and Biodiversity Consortium, and the Hotspots Fire Project in NSW. Some of the ideas outlined here were developed during doctoral work at the University of Western Sydney under the supervision of Ross Bradstock and Charles Morris. Thanks go to all.

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AUTHOR PROFILE

In 2005 Penny Watson completed a PhD on the effects of fire frequency in Western Sydney's grassy woodlands. This followed a Masters degree at Griffith University where she investigated the influence of fire in shrubby woodlands at Girraween National Park near Stanthorpe. Penny has worked as Project Coordinator for the South-east Queensland Fire and Biodiversity Consortium, and as the ecologist with the Hotspots Fire Project, an educational initiative which assists rural landholders in NSW to manage fire for conservation outcomes. In 2008 she joined the University of Wollongong's Centre for Environmental Risk Management of Bushfires, to investigate variability in fuels across time and space.

PRESCRIBED BURNING IN CATALONIA: FIRE MANAGEMENT AND RESEARCH

E. PASTOR, Y. PÉREZ, M. MIRALLES & E. PLANAS

Pastor E., Pérez Y., Miralles M. and Planas E. 2008 06 25: Prescribed Burning In Catalonia: Fire Management And Research. *Proceedings of the Royal Society of Queensland*, 115: 23-27. Brisbane. ISSN 0080-469X.

Wildfires have been seriously affecting Catalonia (NE Spain) during the last decades. Before the large fires of 1994 and 1998, the Catalonian policy was mainly focused on fire extinction activities, underestimating prevention and forest management. Nevertheless, since the occurrence of those events, a new debate arose between politicians, the scientific community and land managers in order to find solutions to the new large wildfire reality. Its first result was the implementation of a new policy based on fire management, basically focused on prescribed burning, as well as the increase of Catalan universities involvement in forest fire research topics. The most relevant achievements in prescribed burning during its six years of application will be presented in this paper, deserving special attention to the synergies found between forestry management and research in wildfires.

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BACKGROUND TO THE PROBLEM OF WILDFIRES IN CATALONIA

Catalonia (NE Spain), as many other regions of the Mediterranean Basin, has been devastated by wildfires during the last decades. Some recent statistics in terms of number of fires and affected area show the magnitude of the current problem

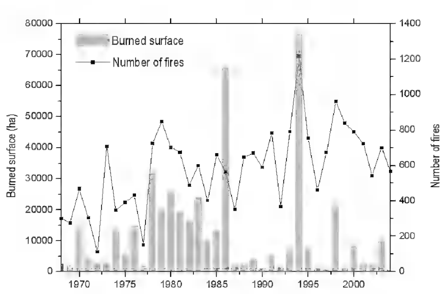

FIG. 1. Number of fires and burned area in Catalonia (1968-2004). Source: Department of the Interior, Generalitat de Catalunya.

Small and medium fires were the most frequent ones during the seventies and early eighties. Nevertheless, the proportion of burned area by these types of fires has decreased to the detriment of large fires. This tendency could be fully studied in terms of the controversial paradox extinction theory (Minnich, 1983; Minnich & Chou, 1997), as even Catalan fire fighters are very successful in 96% of the arisen fires, stopping them from spreading more than 10ha. They were unable to control the remaining 4% of fires, and these are responsible of the 96% of the total area burned per year (Castellnou *et al.*, 2005).

The causal framework of this situation is obviously complex and involves many points such as political, ecological and social items. However, it can be summarised as two main evidences, which are basically the increase of both ignition and propagation risk. The former has its main origin not only in severe meteorological summer conditions but in the increasing social pressure within the forest-urban interface area, due to high demand for nature spaces for recreational uses and accommodation. The later is a direct consequence of biomass accumulation in Catalan forests, mainly due to a) the failure of the primary sector – leading to rural areas abandonment and

lack of forest exploitation and b) to the forest policies historically focused on extinction of fires instead of their prevention, by which small and medium fires acting as natural landscape breaks have practically disappeared. Many resources were allocated during the second half of the 20th century to fire extinction systems in order to increase suppression effectiveness and detection, and so trying to completely remove fire from Mediterranean ecosystem. This policy was found to be hardly useful during the large fires of 1994 and 1998 so, after the occurrence of those dramatic events, a new debate about fire management, essentially in terms of prescribed burning and prevention, finally arose between politician, scientific community and land managers in order to find solutions to the new wildfire reality.

FIRE MANAGEMENT IN CATALONIA. FIRST STEPS

The establishment of this new fire policy in Catalonia was done with the general aim of recovering and reintroducing fire in the country, as it has historically had an important role in Mediterranean ecosystem dynamics, especially relevant in our region of study. This new scope basically meant the introduction of prescribed burning in the ecosystem as well as in the Catalan society with the aim of dealing with the problem of large fires. An ambitious program was established considering three main milestones, i.e. burns design, execution and education. So, bearing in mind these points, prescribed fire has been gradually applied during the last six years, overcoming several difficulties such as bureaucracy, available funds and staff, and public suspicion.

Moreover, early steps in other fire management directions have already been done. At present, some fires are not simply extinguished but managed by monitoring the fire to minimize the simultaneity or to stop it only in anchored sites, directing large fires to minimize potential or avoid vulnerable sites are examples of current fire management decisions in real emergencies situations. Finally, Catalan forest fire fighters are preparing themselves to start managing lighting fires, as this type of fire has traditionally played an important role in fuel discontinuity generation, providing natural sources of biodiversity (Castellnou *et al.*, 2005).

PRESCRIBED BURNING AS A FIRE MANAGEMENT TOOL AND A SCIENTIFIC RESEARCH SCENARIO

CURRENT PLANNING OF PRESCRIBED BURNING. TYPES, OBJECTIVES AND METHODS

Following the mentioned background, prescribed burn planning in Catalonia is done by the Forest Actions Support Group of the Autonomous Government (GRAF, Generalitat de Catalunya) by considering several needs and aims of different natures and priority. These include fire extinction, forest management, training, education and research in different types of forest structures. Hence, prescribed fires may be grouped into two broad categories; the first is in the scope of fire extinction purposes and gathers treatments to transform a forest structure that can be safely used to anchor fire operations, to reduce fuel load in order to decrease fire intensity or to specially protect the rural-urban interface from wildfires. The latter is in the field of forest management and gathers different purposes such as pasture maintenance, thinning or habitat recovery. In that sense, GRAF defines and executes the next types of burns:

- Strategic plots (SP): This category applies to plots where a specific forest structure is manipulated, so that fire fighters can plan secure and efficient manoeuvres in potential large fires. This is especially important when the use of technical fire during fire extinction is needed, i.e. when anchor safer points are required. Location, shape and dimensions of the plot are designed by studying the local fire pattern and regimes, considering historical fires and reproducing their behaviour by means of forest fire simulation tools such as Farsite® and Flammap® (Martínez *et al.,* 2005). The treatment of these critical spots aims to achieve the optimum value of the treated/protected forest surface ratio.

- Low fuel load lines (LFL): This type of burn covers all the actions mainly done next to trails, roads and firebreaks, planned to reduce fuel load and therefore fire intensity, so that the security of the means of fire extinction can be better guaranteed during fire emergencies.

- Interface areas (IA): This group includes burns in all the different types of interface that can be found in Catalonia, i.e. the metropolitan

perimeter, made of sites and fields close to industrial and urban areas where intentional and negligent ignitions may often occur during fire season; the rural interface, where small mountain villages surrounded by abandoned lands must be protected, and the last type of interface area, which has been growing without any control during the last ten years, is the one located in new second-housing developments in the middle of the forest, especially in these areas where forest management has been absent.

- Forest management (FM): They cover different aims and objectives such as thinning, logging debris elimination, habitat recovery for protected fauna and flora species and grazing lands maintenance in the mountains, invaded at present by shrub lands, where used for pasturage before the emigration of rural population to urban areas

PRESCRIBED BURNING AS A FIRE SCENARIO FOR RESEARCH PURPOSES

During recent years in Catalonia, many research teams from Catalan universities have focused their interests in forest fires research topics, joining efforts, sharing resources and working with a close interaction by means of a well defined thematic network funded by the Catalan Autonomous Government (Plana, 2004). Is this framework Catalan fire researchers have found prescribed burning to be a hot topic for their research projects. They cover different ecology subjects such as fire effects on soil properties and on vegetation dynamics. For example, Úbeda *et al.* (2005) have recently examined the immediate and one year effects in soil quality of a low intensity prescribed fire in grassland in terms of pH and nutrients, while Nebot (2005) has compiled in a shared database the information generated during prescribed fires in terms of execution, general fire spread patterns and fire meteorology, so that dynamics vegetation and fire regime have been properly modelled (Piñol *et al.*, 2005).

Moreover, fire behaviour is also studied in prescribed burns, since field fire data is needed for model development and calibration. Hence, Perez *et al.* (2005) have described the field methodology used for IR measurements during a prescribed burn mainly designed for fire extinction purposes and showed their primary results. Finally, other disciplines such as social sciences studies have found in prescribed burns a source of research information, particularly of public perception of this fire use. The studies of Oliveras (2005) and Bell & Oliveras (2006) show

FIG. 2. Recent prescribed burning in pasture lands (Prats de Molló, Pirineus, Catalonia). Source: Department of the Interior, Generalitat de Catalunya.

the advantages and drawbacks of prescribed burning perceived by two sample populations, the former in Catalonia and the latter in central western Victoria, comparing both areas in terms of knowledge, participation and acceptance.

RESULTS AND DISCUSSION

During this six years of prescribed fires in Catalonia the main goals have been successfully achieved (Nebot, 2005). It has become a normalised tool for fire prevention as the area treated by prescribed fire has been increasing year after year, although the real impact of any prescribed burning program on fire statistics (e.g. number of large fires) will take many years to be assessed.

This empirical evidence has been endorsed by Piñol *et al.* (2005) by means of their model of vegetation dynamics and fire spread. Hence the authors found a clear relationship between the increase of prescribed burn intensity and the decrease of large fire occurrence in the regions of study (South Catalonia and Central Portugal). Nevertheless, despite the potential of this tool, prescribed burning is a big source of controversy in Catalonia. Thus, a great debate has currently ensued between politicians, land managers and the public (Oliveras, 2005) with escape risk, public security, inconveniences caused by smoke, reduction in air quality, associated aesthetic issues and fiscal responsibilities being the main concerns.

The debate is still open, not only in social terms but in the ecological effects. Thus, wider studies are needed in order to better assess the entire repercussions of prescribed burning in Mediterranean ecosystems – particularly in Catalonia – as well as the specific efficiency of this tool in the achievement of all the different planned objectives. Practical and useful methodologies have to be designed in order to obtain this valuable information. Finally, the productivity and profitability of prescribed burning as opposed to other treatments has to be discussed (Larrañaga *et al.*, 2005).

Furthermore, more agreement and communication is still needed between researchers and fire managers in order to completely achieve burn objectives, to complement methodologies and to optimize resources, though much effort has been already invested in this direction.

CONCLUSIONS

Prescribed fire has been presented as a valid tool in the early stage of its application for fire management in Catalonia. Although more research is needed in order to better understand long-term ecological effects and efficiency in terms of forest management objectives, the Catalan research community is already working on these topics.

Prescribed burning represents a hot forest fire research topic in Catalonia. An emergent synergy between forestry management and researchers from multiple disciplines has been recently found. Prescribed burning with preventive objectives offers an inestimable opportunity to develop experimental scientific studies encouraging the interaction between both research and operational staff.

Prescribed burns can be optimised with other aims that cover not only fuel management and extinction purposes but research needs in several areas, promoting advances in both directions. Because of this great potential, an increment of the efforts necessary to normalise this practice is strongly demanded.

ACKNOWLEDGEMENTS

To the Autonomous Government of Catalonia (Generalitat de Catalunya). This work is sponsored by the Ministry of Science and Technology Comission (Project AGL2005-07269/FOR).

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PRESCRIBED BURNING IN THE SOUTHERN MT. LOFTY RANGES: HOW AND WHY IS THE DECISION TO BURN MADE?

NOEL W RICHARDS

Richards N.W. 2008 06 25: Prescribed Burning In The Southern Mt. Lofty Ranges: How And Why Is The Decision To Burn Made? *Proceedings of the Royal Society of Queensland*, 115: 29-35. Brisbane. ISSN 0080-469X.

Preliminary results are presented from a case study being conducted in the Southern Mt. Lofty Ranges region near Adelaide. This region is characterised by high bushfire risk, increasing land use fragmentation and areas of remnant vegetation of conservation significance.

Prescribed burning decisions in the region are made within a context where values associated with the reduction of bushfire risk to human life and assets, may conflict with values associated with biodiversity conservation. Decisions are also made in the legislative context of state Acts which may either compel bushfire hazard reduction activities, such as prescribed burning, or conversely may prohibit the burning of native vegetation.

To understand the 'how and why' of prescribed burning decisions a qualitative methodology is used with reference to decision theories, in particular 'bounded rationality' and 'disjointed incrementalism'. To date the research has found that prescribed burning decisions were, until recently, made on an ad hoc basis on most land tenures. This is changing, particularly on public lands, with the formation of inter-government bodies to coordinate and codify the process. Nonetheless, it is evident that political imperatives play an important part in the decision and are arguably a greater influence than empirical knowledge.

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Gill *et al.* (2002) identified that the values and ideas associated with fire management have largely gone undocumented in the scientific literature. This paper presents the preliminary results of research being undertaken to determine what values and ideas influence the 'how and why' of prescribed burning decisions and thereby contribute to filling the gap identified by Gill *et al.* (2002). Prescribed burning has attracted controversy in Australia since the 1960s with the debate characterised by concerns over the ecological effects of burning and the effectiveness of the process in achieving bushfire hazard reduction aims (Gill & Bradstock, 1994; Lindenmayer & Burgman, 2005). Those charged with the responsibility of making prescribed burning decisions face the difficult task of reconciling potentially competing objectives of bushfire hazard reduction and biodiversity conservation. This is especially the case where human life and asset protection is juxtaposed with objectives to conserve indigenous biodiversity, such as in peri-urban environments (Whelan & Baker, 1998; Whelan, 2002; Whittaker & Mercer, 2004).

The results presented are from a case study being conducted in the Southern Mt. Lofty Ranges region near Adelaide in South Australia. The data discussed was collected from June to December 2005. This is the first of two case studies being undertaken with the second to be conducted in the Mt. Macedon region of Victoria in 2006. The Southern Mt. Lofty Ranges region is characterised by a temperate climate, areas of remnant vegetation (native vegetation accounts for an estimated 13 per cent of land cover), hilly topography, a recent history of urban and rural development and land use fragmentation (MLRIINRM, 2003). The region has experienced numerous significant bushfire events and the entire region is zoned as either an extreme or a high bushfire risk for development purposes under the South Australian Development Act 1993. The region is also listed by the Commonwealth Department of Environment and Heritage as one of the 15 'Biodiversity Hotspots' in Australia, as it supports the largest remnants of woodland vegetation communities in South Australia (Dept. of the Environment and Heritage, 2005). The ultimate aims of this research are to present a descriptive theory of

how and why prescribed burning decisions are made and suggest a normative theory on how these decisions may be improved.

METHODS

A case study methodology has been chosen for this work because of its capacity to deal with a full variety of evidence including documents, artefacts, interviews and observations (Yin, 2003). The value of this type of research strategy is that it can discover connections that are not obvious and allow the posing of difficult questions (Flyvbjerg, 2001). The case study approach, employing qualitative methods, is particularly useful to address explanatory 'how and why' questions in the 'real-life' context in which they exist (Stake, 1995; Yin, 2003). The methods employed for this research are semi-structured interviews, participant observation and documentary reviews. At the time of writing, 25 interviews have been conducted with a range of stakeholders (Table 1), these and participant observation, form the primary data collection process for the research. A further 20 interviews are expected to be conducted to complete data collection for this case study.

Secondary data collection has occurred through unstructured key informant interviews and the review of organisational documents such as operational guidelines, guides to regulations and fire management plans. Tertiary data collection has included the review of publicly available documents, relevant legislation and reports. This wide range of data collection enables a triangulation process along converging lines of inquiry and has provided construct validity (Morrison, 2004). Construct validity, a term coined by Yin (2003), which refers to research tactics such as the use of multiple sources of evidence, and the establishment of chains of evidence to enhance the validity of research outcomes.

THEORETICAL POSITION

This research will consider the prescribed burning decision as it relates to four theoretical areas; ecophilosophy, fire behaviour, fire ecology and decision theory. Prescribed burning decisions, and the values they contain, are posited within overarching ecophilosophical positions. Values may be identified which are located within a spectrum of philosophies ranging from 'deep ecology', which would privilege non-human nature ahead of human needs to 'anthropocentrism' where human needs would be privileged above non-human nature. The demarcation between different ecophilosophies however may not always be clear (see for example Hay, 2002).

Prescribed burning decisions are explicitly informed by established fire behaviour science and a growing, but far from complete, body of empirical knowledge about fire ecology. There is considerable empirical and anecdotal support for the view that prescribed burning for hazard reduction purposes is necessary and effective at achieving its objective, although there are caveats concerning site specificity, frequency and spatial arrangements in the prescription design. The recent inquiry commissioned by the Council Of Australian Governments into bushfire mitigation and management gave clear support for hazard reduction

TABLE 1. Interview matrix, Southern Mt. Lofty Ranges case study 2005-2006 (Data codes in TABLE 1 are referred to in the text)

Affiliation	Entity Represented							
	Government			NGOs			Individual Actors	
	Commonwealth	State	Local	NRM	Environment	Indigenous		Data Code
Department for Environment & Heritage		3						DEH 1-3
Commonwealth DEH	1							CDEH
Nature Foundation SA					1			NFSA
Current Native Vegetation Council		1						CNVC
Retired Native Vegetation Council							1	RNVC
Aboriginal Community						1		AC
SACFS		3						CFS1-3
Local Govt.			7					LGA1-7
Retired SA Forestry Corporation							1	RFSA
Planning SA		1						PSA
SA Indigenous Flora							1	SAIF
SA Museum							1	SAM
Urban Forest Biodiversity Program					1			UFBP
Fire Prevention Officer				1				FPO
Friends of Park Group					1			FOP
Totals 25	1	8	7	1	3	1	4	

burning, whilst recognising the complexities of the process and its execution (Ellis *et al.*, 2004). There is also a significant literature that identifies potentially undesirable consequences of frequent hazard reduction burning for vulnerable species (for example Keith, 1996; Morrison *et al.*, 1996; Cremer, 2004). Some regard hazard reduction burning and biodiversity conservation as incompatible; others however, (for example Bradstock and Gill, 2001; Wouters, 2005) articulate approaches based on landscape zoning, assessment of ecosystem functional groups and plant vital attributes as guides for prescriptions to accommodate both objectives.

This research will also consider the prescribed burning decision in the context of the body of literature dealing with decision theory. The works of Simon (1957; 1982) and his concept of 'bounded rationality' with Braybrooke and Lindblom (1963) and their work on 'disjointed incrementalism' are seen as useful frameworks against which to evaluate the decision process. The essence of Simon's bounded rationality is that humans cannot possibly know and evaluate all information and all options that exist when making a decision, in other words we cannot make a comprehensively rational choice. This does not mean that our decisions are necessarily irrational, but rather, that our rationality is limited or bounded by, amongst other factors, our intellectual capacities, values and organisational environments. Braybrooke and Lindblom (1963) accept bounded rationality but are hostile to rationalist views that decision making ought to be a neat step by step process, progressing from a definition of goals through to selection of alternative actions and evaluation of options. They contend that this is simply not workable for complex questions, a category within which prescribed burning decisions comfortably fit. They identify a need for a process that involves a continuum of building out from current knowledge, proceeding by incremental change using trial and error, the foundations of adaptive management perhaps, but labelled by them as 'disjointed incrementalism'. They also contend that decisions need to be based on a degree of stakeholder agreement. This is applicable in the context of prescribed burning, particularly where objectives for land management may clash, such as in the peri-urban realm where collaboration between disparate parties is needed to sustain land management (for example, Boura, 1994). Agreement between parties, politics in other words, is an essential component of fire management, as Pyne (2006) asserts, 'In the realm of bushfires, politics is not an intervention or a distortion. It is the fundamental arena for deciding what should be done and how to do it.'

RESULTS

LEGISLATIVE FRAMEWORK

The key legislative framework that guides prescribed burning in the region, over various land tenures, is outlined in Table 2, together with the principle agencies that have a role in prescribed burning.

The South Australian Country Fire Service (SACFS) is the key agency, outside the metropolitan area, that may conduct prescribed burning on private lands. Prescribed burning may arise as a result of a landholder request, or the identification of what may be regarded as high bushfire risk fuel loads through the operations of district or regional Bushfire Prevention Committees which are constituted under the Fire and Emergency Services Act, 2005 (previously the Country Fires Act, 1989). These committees are comprised of a range of personnel from local and state government agencies and the community. On public lands the Department for Environment and Heritage (DEH), South Australian Forestry Corporation (SAF) and SA Water are the most relevant organisations. Under

TABLE 2. Relationship between responsible agency, land tenure and legislation, Mt. Lofty Ranges 2006

Legislation	**Agency**	**Land tenure jurisdiction (within case region)**
Fire & Emergency Services Act 2005	South Australian Country Fire Service	All private (but has certain powers in respect of all non-metropolitan land)
Native Vegetation Act 1991	Native Vegetation Council	All public & private in respect of planned burning (clearance) of native vegetation
National Parks & Wildlife Act 1972	Department for Environment & Heritage	National parks and conservation reserves
Forestry Act 1950	South Australian Forestry Corporation	Forest plantations and native vegetation reserves
South Australian Water Corporation Act 1994	S A Water	Reservoir and water infrastructure lands

the Native Vegetation Act, 1991 burning of native vegetation stratum is regarded as clearance requiring the approval of Native Vegetation Council (NVC). There are some exceptions relating to asset buffers, access roads and emergency situations. This approval applies to all public and private land tenure although authority has been delegated to DEH to conduct its own assessments, subject to annual review by the NVC, except for wilderness protection areas.

PRESCRIBED BURNING CASE STUDY BACKGROUND

Interview participants widely acknowledged that South Australia is in the early stages of using prescribed burning as an environmental management tool whilst its use in the case study region for bushfire hazard reduction has only re-emerged in the last 2 years. This follows a decade or so of reluctance to employ it due to environmental concerns (Interview DEH 3). The SAF was cited as the agency with the longest active experience in prescribed burning, both as part of silvicultural practice and as part of management of its own native vegetation reserves, predominately as a means of reducing bushfire hazard.

The re-emergence of prescribed burning was attributed, in part, to the 'Bushfire Summit' (the Summit) of 2003 convened by the South Australian Government to examine a number of bushfire related issues including 'the desirability of a burn off program of excess fuel loads…' (Government of South Australia, 2003). The Summit was held following the Canberra bushfires of January 2003 and was used to announce a budget increase of $21M to enhance bushfire prevention and suppression capacities in the state. Within DEH, the agency responsible for the management of the State's national parks and conservation reserves, additional funds were provided to create a dedicated fire management section to increase fire management capacity within national parks. The development of prescribed burning guidelines and policy being an important component of the section's work. The Summit, and the investment in fire management that accompanied it, created what was widely recognised as a 'political imperative' for evidence of action in the field, as a Friends of Parks member recalled, with reference to fuel reduction burning, 'it came down from the Minister that there had to be some burns'.

VALUES AND IDEAS EMERGING FROM THE DATA

The Summit also called for the streamlining of clearance approvals required under the *Native Vegetation Act 1991*. This issue was also on the minds of SACFS officers who, during interviews, were critical of the NVC, who administers the Native Vegetation Act 1991, for being too slow in approving applications to conduct prescribed burning on private land (Interviews SACFS 1, 2 &3). They were also concerned that the NVC was being unreasonable in expecting landholders to justify that proposed burns would not cause undue damage. The concerns held by SACFS officers are perhaps an inevitable consequence of the different values that inform the operational objectives of the SACFS and the NVC. Values institutionalised in the legislation under which each organisation is constituted. The SACFS is required under the *Fire and Emergency Services Act 2005* to 'provide services with a view to preventing the outbreak of fires, or reducing the impact of fires' (Division 2, 59 (1) a). In pursuit of this duty the SACFS regard prescribed burning for bushfire hazard reduction as an effective means, amongst others, of preventing bushfire occurrence and severity (Interview SACFS 1, 2 & 3).

A key object of the *Native Vegetation Act 1991* is 'conservation, protection and enhancement of the native vegetation of the state and in particular, remnant native vegetation' (Part 2, 6 (a)), and burning is regarded by the Act as 'clearance' of native vegetation. *The Fire and Emergency Services ACT 2005* does not disregard the environment, indeed protection of environmental assets is an expected function of the SACFS under this Act. Nor does the *Native Vegetation Act 1991* or the NVC disregard the need for reasonable measures to protect human life and assets. That it is divergent values that are at the heart of this issue is made clearer by the limitations imposed by time and finite resources.

A thorough assessment of prescribed burning impacts on native biota at a given site takes considerable time and expertise to assemble and analyze. However the bushfire threat that high fuel loads may represent is, within the bushfire season, more or less immediate. If high fuel load risks are being identified only as they peak, rather than being predicted, then the pressure to address that risk becomes immediate and the values of conservation and the protection of human life and assets are more likely to clash.

Interviews with those for whom conservation of indigenous biodiversity is a key objective revealed that the concept of applying pre-European fire regimes was broadly accepted as an important process for conserving indigenous biodiversity (Interview UFBP; FOP). This was associated with an awareness that indigenous biodiversity was a product of millennia of Aboriginal fire use and that analogous regimes would be required to achieve conservation goals. The return to, or maintenance of pre-European vegetation communities, was cited as an ideal of conservation efforts (Interview FOP; UFBP). The pre-European ideal appeared to be regarded as synonymous with 'natural' even though Aboriginal fire use has created what is, in effect, a cultural landscape. Wouters (2005) on the other hand argues that 'too much emphasis has been placed on obtaining an assessment of 'natural' fire regimes' which he regards as probably impossible to determine.

The prescribed burning debate in Australia has long involved a contention that burning for hazard reduction reasons is incompatible with ecological fire regimes (Gill & Bradstock, 1994). This remains contentious in the case study region. In interviews with SACFS and DEH staff, hazard reduction objectives and biodiversity conservation were regarded as mutually achievable outcomes, provided appropriate planning and assessment is undertaken (Interviews SACFS 1, 2 &3; DEH 2 &3). However the view expressed by Vickery (2005) that 'It is illogical to suggest that in the Australian context, we can remove fuel without reducing biodiversity' reflects concerns that continue to be held in conservation circles in the region. Various interview participants also raised the concern that not enough research had been done in the region to make informed decisions and that research findings from other states were not necessarily applicable in the Mt. Lofty Ranges (Interview CNVC; NF; SAIF). Certainly the region is not a homogenous environment and as the Nature Foundation representative cautioned 'simplistic answers were to be avoided and it is not possible to apply the results of research from elsewhere in Australia'.

The established view that fuel reduction burning could be effective at achieving its purpose was challenged by a research participant who asserted that there is evidence that elevated fuel loads in the region's stringy bark forests (*Eucalyptus obliqua* and *E. baxteri*) actually reduce in the long term absence of fire. This was attributed to the senescence of mid-layer vegetation such as *Acacia pycnantha*. The informant held that mid-layer fuel loads, in these forests, were actually increased by fire frequencies of between 5 and 10 years, as may be applied with hazard reduction burning, due to fire induced generation of many mid layer species (Interview SAIF).

The debate over prescribed burning embodies broader philosophical questions of humanity's engagement with the non-human environment, how nature is perceived and what this means for natural resource management. The desire to conserve native vegetation, and the ecosystems it sustains, and the desire to protect human life and assets are both widely held by the Australian community and are manifest in the legislation that has been discussed here. These desires are more likely to become mutually exclusive when pressure to act compromises a planned and considered approach.

Parsons (1995) in discussing decision analysis and Simon's 'bounded rationality', suggests that decision analysis is not about ensuring that correct decisions are made, but rather, that a good decision making process is followed. This research does not aim to develop a case for or against prescribed burning per se, rather it is about critically investigating the decision process so that possibilities for its improvement may be suggested. A process better able to reconcile the divergent values that exist.

DISCUSSION

It is evident that prescribed burning within the case study region has until recently, with the exception of SAF lands, been a relatively infrequent activity. However following the stimulus of the Bushfire Summit, conducted in response to the Canberra 2003 fires, prescribed burning activity for hazard reduction reasons has been increasing on public lands, particularly national parks and reserves managed by DEH. The implementation of burning on these land tenures has occurred following investment in human resources to assess and develop a planned and codified approach which at the time of writing is continuing to be developed. On private lands however an overarching landscape scale approach is not yet evident, although its attainment remains a goal. The complexities of landscape scale planning across fragmented private land tenure will require an effective prescribed burning decision making process, one that deals with divergent values and ideas.

ACKNOWLEDGEMENTS

This work is conducted with the financial assistance of an Australian Postgraduate Award and the Faculty of Social Sciences at the Flinders University of South Australia. The support of my supervisors, Dr. Tiffany Morrison, Dr. David Bass and Professor Iain Hay and of all those who have participated in this research is gratefully acknowledged.

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CHANGES TO SIMULATED FIRE BEHAVIOUR AND SOCIETAL BENEFITS AFTER TWO LEVELS OF THINNING IN A MIXED-CONIFER WILDLAND-URBAN INTERFACE COMMUNITY

CHRISTOPHER A. DICUS

Dicus, C.A. 2008 06 25: Changes to Simulated Fire Behaviour and Societal Benefits after Two Levels of Thinning in a Mixed-Conifer Wildland-Urban Interface Community. *Proceedings of the Royal Society of Queensland*, 115: 37-44. Brisbane. ISSN 0080-469X.

Potential fire behaviour and various societal benefits (air pollution removal, carbon sequestration, and stormwater runoff) were quantified in a California Sierra mixed-conifer forest in (a) untreated conditions, (b) after removing all understorey trees <15 cm dbh, and (c) after thinning 50% of the stand's total basal area. Potential fire behaviour was modelled under constant conditions near a hypothetical development by the FARSITE fire behaviour and growth simulator and societal benefits were calculated by CITYgreen, both GIS-based software applications. Results showed that fire behaviour was considerably moderated by both thinning treatments. Modelled societal benefits, however, were largely unaffected by either treatment, which may be the result of inherent assumptions in the model. Critical elements of sustainable development in the wildland-urban interface are discussed, including fuels management, enforceable construction standards, sound land-use planning, community education, and appropriate suppression resources. Each of these components will vary depending on the ecosystem and socioeconomic conditions of a given area that is under consideration.

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California, more than any other region of the United States, illustrates the increasing challenges of fire management in the wildland-urban interface (WUI). A burgeoning population who lives amongst a menagerie of volatile vegetation types is increasingly leading to extensive costs and losses associated with wildfires. Vegetation in California is largely a product of a Mediterranean climate, where a mild rainy season is followed by periods of up to 8 months of drought. Therefore, both live and dead fuel moistures are regularly at critical levels for extreme fire behaviour. Additionally, high-intensity foehn winds are common on the central and southern coasts of California during the driest months of the year, contributing to the size and intensity of fires in the area. Further, in many montane forests in California and throughout the western United States, a century of fire exclusion has led to vast levels of overstocking, leading to extremely high fuel loading and continuity.

Fourteen fires in California have burned over 40,000 ha, including the 2003 Cedar Fire in San Diego County, which burned over 105,000 ha (California Department of Forestry & Fire Protection, 2007a). While similar sized fires may be experienced elsewhere, the immense population in California, currently 35 million and expected to grow to 46 million by 2030 (California Department of Finance, 2004), leads to losses unlike most parts of the world. For example, 14 fires have consumed over 300 homes there, again led by the Cedar Fire, which consumed 4847 structures (California Department of Forestry & Fire Protection, 2007b). Of greatest concern is that 7 of the 9 most destructive fires in California have occurred in the last 20 years. Given demographic trends of continued immigration to wildland areas, most feel that the trend of highly destructive fires will continue largely unabated in the foreseeable future.

In addition to immediate fire effects, wildfires in California are regularly followed by mudslides and immense sediment deposition into streams and estuaries, which significantly degrade public safety and environmental quality. For example, a mudslide following the 2003 Old Fire on the San Bernardino National Forest killed 14 people. And the 1994 Highway-41 fire on the Los Padres National Forest caused accelerated erosion into the Morro Bay National Estuary, leading to lowered taxonomic diversity there for several years (U.S. Environmental Protection Agency, 1995).

Therefore, it is imperative that agencies tasked with wildland fire management develop efficient and effective management strategies in the WUI, which should vary by site dependant on the biophysical and socio-political factors present. Unfortunately, it seems that many fire managers in the United States regularly think of vegetation simply in terms of fuels and potential fire behaviour, overlooking the many tangible benefits that vegetation provides. Such societal benefits in the WUI could include reduced home cooling costs and air pollution (Taha *et al.,* 1997), lessened need for stormwater runoff infrastructure (Sanders 1986), increased carbon sequestration (Rowntree and Nowak, 1991), wildlife habitat, and others. Thus, to best insure sustainable development in the wildland-urban interface, site-specific vegetation management plans must be developed that minimises fire risk while simultaneously maximizing the benefits that distinct vegetation communities provide.

Currently, no such single instrument exists to assist fire managers in developing fuel modification prescriptions to maximise both elements. However, existing software packages that examine each element individually could be used to facilitate effective and environmentally responsible prescriptions. Fire behaviour modelling programs such as BehavePlus (for surface fire behaviour), NEXUS (for crown fire potential), and FARSITE (for fire spread across landscapes) could be used in conjunction with CITYgreen or other programs that quantify specific societal benefits. The results could then be used to derive a plan that simultaneously minimises fire behaviour while maximising benefits that vegetation bestows.

The principal objective of this manuscript is to illustrate how various treatments simultaneously affect fire behaviour and societal benefits using two GIS-based software packages commonly used in the United States. Specifically, the author explores how modelled fire behaviour and various societal benefits change in a California Sierra Nevada mixed-conifer forest after two levels of thinning intensity. Further, elements critical to sustainable development in the WUI are explored. While centric to California, it is hoped that the successes and lessons learned there can be applied to similar regions of the world.

MATERIALS AND METHODS

A hypothetical WUI community was created to illustrate the potential effects of thinning on fire behaviour (Fig. 1). In the hypothetical scenario, a common ignition point for all fire behaviour simulations was located below a subdivision of homes, which set higher atop a ridge. The untreated landscape largely consisted of dense, mixed-species stands of conifers having a high degree of vertical continuity, which would likely facilitate torching and active crown fire spread via spotting. Indeed, 95% of the surface fuel models across the landscape consisted of dry climate timber-shrub types (Timber Understorey models per Scott & Burgan, 2005). Two fuel treatments were implemented across the landscape, including thinning all trees under 15 cm (Understorey Removal), and thinning-from-below to 50% of the total stand basal area (Thin to 50% BA). For each fire behaviour simulation, all inputs were held constant except canopy base height and canopy bulk density, the values of which were input across the landscape for a given simulation per Scott and Reinhardt (2005) for identical treatments in Sierra Nevada Mixed Conifer (Table 1).

Assuming constant conditions (Table 2), FARSITE (Fire Area Simulator v. 4.1.03) was utilised to model fire spread for 10 hours from a single, common ignition point before and after each treatment was implemented throughout the landscape. Community benefits of air pollution removal (ozone, SO^2, NO^2, CO, and particulate matter), carbon sequestration, and stormwater storage capacity were calculated by CITYgreen for ArcGIS. CITYgreen vegetation classifications were input across the landscape by converting fuel model raster data utilised by FARSITE to an appropriate CITYgreen "feature". In all simulations, the grassland areas were classified in CITYgreen as "Open Space - Grass/Scattered Trees>>Grass cover > 75%". The forested areas were classified as "Trees>>Forest litter understorey>>No grazing, forest litter and brush adequately cover soil" in the untreated landscape, "Trees>>Forest litter understorey>>Grazed but not burned, some forest litter" after the Understorey Removal treatment, and "Trees>>Forest litter understorey>>Forest litter and brush destroyed by grazing or burning" after the Thin to 50% BA Treatment.

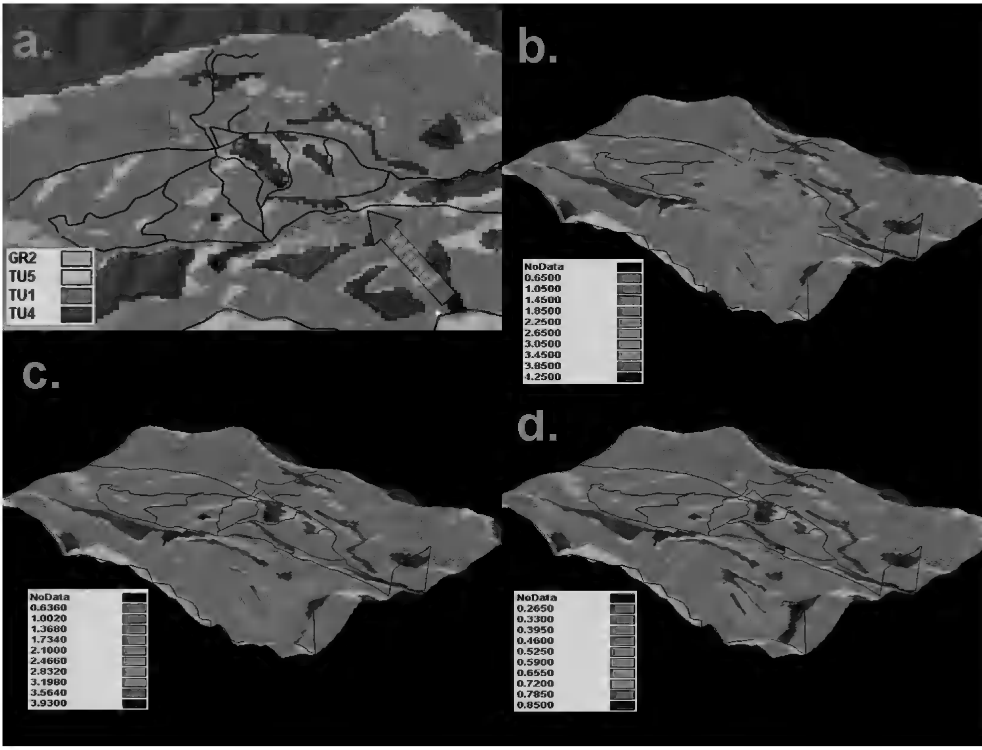

FIG. 1. [a] Wildland-urban interface community in a Sierra Nevada mixed-conifer forest (yellow squares = houses), and simulated fire boundary and flame length (m) after 10 hours [b] without treatment, [c] after understorey removed (<15 cm dbh), and [d] thinned to 50% basal area. Varying colors across landscape designate fuel models (see inset in 'a' for specific fuel models per Scott and Burgan (2005)). Varying colors within fire boundaries designate flame length (see inset in 'b'-'d' for specific flame lengths).

TABLE 1. Canopy characteristics of a Sierra Nevada mixed-conifer forest in an untreated landscape, after removing all understorey trees <15 cm, and after thinning from below to 50% of the initial stand basal area.

Simulation	Stand ht (m)	Canopy Base ht (m)	Canopy bulk density (kg/m^3)
Untreated	34	2	0.101
Understorey Removal	34	4	0.101
Thin to 50% BA	33	11	0.037

RESULTS

Both flame length and area burned were substantially reduced during the 10-hour simulation after both thinning treatments (Fig. 1). In the untreated landscape, the simulated fire burned 203.5 ha, but was reduced to 76.3 ha in the Understorey Removal treatment and 63.9 ha in the Thin to 50% BA treatment.

While thinning substantially impacted simulated fire behaviour, the calculated societal benefits were largely unaffected by either treatment (Table 3). Indeed, there were no detectable differences in either pollution removal (179 t for all simulations) or carbon sequestration (1,294 t for all simulations) between any of the scenarios. Further, stormwater storage capacity changed only slightly between treatments, ranging from 219,768 m^3 in the untreated landscape to 170,754 m^3 in the Thin to 50% BA treatment.

DISCUSSION

Both thinning treatments considerably reduced simulated spread rate and flame length in the untreated landscape. This reduction is apparently the result of an increase in canopy base height (from 2 m to 4 m) and not a reduction in canopy bulk density. Increasing the intensity of thinning had minimal effect on fire spread because the greater canopy base height after either thinning discouraged torching and subsequent lofting of embers ahead of the main fire front, thereby limiting fire spread and intensity. Because few trees torched after raising the canopy base height, further reducing the canopy bulk density in the more intense Thin to 50% BA treatment (from 0.101 kg/m3 to 0.037 kg/m^3) had minimal impact to simulated fire behaviour. Managers, however, might still consider the more intense thinning in order to generate revenue from the sale of larger trees so as to recover costs incurred in a non-commercial Understorey Removal treatment.

Although thinning significantly impacted fire behaviour, the societal benefits derived from the vegetation changed very little. Indeed, the only change to calculated benefits was a slight reduction in the stormwater storage capacity afforded by the vegetation. The absence of significant change seems likely due to an over-reliance on overstory canopy coverage, which is assumed constant for a given vegetation type, as CITYgreen's primary variable when calculating most benefits. Thus, even though total vegetation would obviously decline after thinning, the program does not

TABLE 2. Landscape characteristics and FARSITE inputs used in fire behaviour simulations in a Sierra Nevada mixed-conifer forest.

Weather	**FARSITE Model Parameters**
Temperature: 32°C	Timestep: 30 minutes
Relative humidity: 30%	Perimeter Resolution: 30m
Wind: 35 kmph (SE)	Distance Resolution: 30m
Topography	
Slope: 15-30%	**FARSITE Fire Behaviour Options**
Aligned with wind	Crown Fire: enabled
	Spot Fire Growth: enabled
Fuel moisture (initial)	Ignition Frequency: 1%
Live: 90%	
Dead: 4%	

TABLE 3. Societal benefits calculated in a Sierra Nevada mixed-conifer forest in an untreated landscape, after removing all understorey trees <15 cm, and after thinning from below to 50% of the initial stand basal area.

Simulation	Stormwater capacity (m^3)	Air pollution removal (t)	C sequestered (t)
Untreated	219,768	179	1,294
Understorey Removal	193,468	179	1,294
Thinned to 50% BA	170,754	179	1,294

consider differences in forest structure. Instead, the software classified all forested stands as "Trees" with choices only for the types of ground cover, which did not affect any benefit of interest other than stormwater storage capacity. Dicus & Zimmerman (2007) found that CITYgreen also calculated zero benefits for all non-tree types of vegetative cover, thereby limiting its reliability in sites where forests were not the dominant vegetation on the landscape. Thus, as with all models, users should understand the assumptions and limitations when using this software. New software has been recently developed or modified, which may address some of the challenges experienced in the present and past studies. For example, the Street Tree Management & Analysis Tool (STRATUM) sums the benefits derived from individual trees, but may not be particularly suited for wildland settings (Dicus, 2006). The Urban Forest Effects (UFORE) model, which quantifies species composition, diameter distribution, tree density and other structural characteristics, seems especially promising, particularly because it is also able to calculate benefits derived from brush species (Nowak & Crane, 2000). While subtle, results here indicate that treatment-induced reductions in societal benefits can occur and therefore, fire managers must be cognizant of potential changes to not only fire hazard, but also societal benefits when implementing any fuel treatments

Largely because of the societal and environmental benefits that vegetation provides, proper fuels management is an essential but controversial component of any proper WUI management strategy. Because of the 2003 firestorms in southern California, which ultimately burned over 303,500 ha, destroyed 3710 homes, and killed 24 people, State legislation (California Senate Bill, 1369) now requires that private landowners modify vegetation within 30.48 m of any structure so as to reduce potential fire hazard. There is increasing concern that the increased clearance regulations will further degrade and fragment native vegetation, which has been significantly impacted by burgeoning urban sprawl and development (American Forests, 2003). It is imperative, therefore, for the prudent fire manager to consider both fire hazard and societal benefits when implementing any fuel modification treatment.

There are many fuel treatments available to fire managers, each with positive and negative aspects. WUI fire managers should not take a one-size-fits-all approach to fuel modification, but should consider both the biophysical and sociopolitical factors present in a given area. Treatments currently available to land managers in California include prescribed fire, mechanical mastication, hand piling and burning, chipping, goats and other livestock, herbicides, and others. Prescribed burning is the most opposed tool in the WUI due to the potential for escape and lowered air quality. Even where socially accepted, it is often impossible to use prescribed fire on many sites until some other type of mechanical treatment has been conducted because of dangerously high fuel loadings. While prescribed fire is by far the most cost- and objective-effective means of reducing fuels, it will likely decrease in use in California as the population continues to expand.

Of note, one trait that facilitates the assessment of non fire-related aspects of vegetation is that many fire managers in the United States were educated in forestry or other resource management disciplines. This allows them to see beyond how various treatments will affect fire behaviour and better comprehend the multifaceted effects of vegetation manipulation. Unfortunately, it appears that wildland fire agencies are becoming increasingly more engrossed in emergency services and less in resource management. General consensus is that the fire suppression and resource management aspects in both the federal United States Forest Service and the state California Department of Forestry and Fire Protection are increasingly diverging into their own distinct entities, which will adversely affect the abilities of future fire managers to fully appreciate the non-fire aspects of vegetation management.

Fuel modification, however, is only one aspect of sustainable WUI management. In general, fire managers should also address four additional aspects of WUI management so as to reduce the costs and losses associated with wildfires. These other critical elements include construction standards, sound land-use planning, effective community education, and increasing fire suppression success. While priorities should be established dependant on the local situation, no element should be entirely absent in the management of WUI communities.

Construction standards and sound land-use planning will significantly impact the degree of fuel modification needed in a given area. Individual homes should adhere to adequate, enforceable regulations that

address fire protection in siding, vents, windows, and especially roofs, which are particularly susceptible to ignition from burning embers (Cohen, 2000).

Proper land-use planning that provides for appropriate housing density, home placement, and infrastructure needs, such as ingress/egress and water supply, is also essential (Schwab & Meck, 2005). New development in California must undergo intensive review by local government entities before building permits are issued. If sound land-use principles are not applied in the permit approval phase, then there will be a multiplier effect that will negatively impact the development for many years.

Obviously, older developments that were constructed largely before construction and land-use standards were enacted are usually at much greater risk to wildfire than newer communities where the right to develop was hinged on the ability to adequately address fire concerns. To reduce fire risk in California, even government-issued permits to remodel one's home can trigger mandatory upgrades to fire-resistant construction. For older developments, it is imperative to have adequate codes in place before a fire event and after the event subsequently disallow any new construction that does not meet new construction standards. While many victims of wildland fire have vehemently complained of burdensome government obstruction in rebuilding efforts, improved construction standards will reduce a cycle of repetitive loss. That said, it should be noted that higher construction standards will translate into higher construction costs. Given the high cost of housing in California, where the median home price for the State currently exceeds $535,000 (U.S.), the need for affordable housing and fire standards will inevitably clash.

Effective community education is another critical component of effective fire management in the WUI. Education efforts must be developed for a specific target community based on the level of local knowledge. Managers in the WUI need not assume that community members are totally ignorant of the threat of wildland fire. Some communities are well aware of the threat, but lack specific knowledge on how to reduce their risk. Thus, funds allocated to public service announcements over mass media outlets such as radio may be essentially wasted in those communities. Further, research has shown that while cost-effective and requiring less effort, mass media advertising has little value in effecting change in the behaviour of WUI residents (McCaffrey, 2004). Instead, personal contact has the greatest impact on changed behaviour. However, it is virtually impossible for fire personnel to visit each home in its responsibility area given budgetary and time constraints. However, by organising interactive, informational displays where the public would likely be present, such as at a hardware store or county fair, fire personnel have successfully been able to provide personal contact with community members.

The final component of a proper fire management strategy is properly equipped and staffed fire suppression forces. Having the proper types and numbers of suppression equipment and personnel are essential to adequate fire protection in the WUI. For initial attack success, it is critical that the appropriate resources are in the right place at the right time. In general, a community decides their level of service, which is a measure of the percentage of fires controlled by initial attack, through voter-approved property taxes. It is the responsibility of fire administrators to allocate funds to specific resources within the organisation to best increase success of initial attack. If budgets are managed efficiently and the level of service is unacceptable for a given community, those residents must be willing to increase taxes on their property. However, in many areas it is extremely difficult to convince voters to increase property taxes, which subsequently limits the ability for fire agencies to properly respond to fire events.

Regularly, it seems that the fire suppression aspect of WUI management is overemphasised in the United States by both fire agencies and the public it serves. By first adequately addressing the other four elements of sound WUI management, the demands on the fire suppression community will be lessened and their effectiveness will subsequently be increased. Analogous to the military, when battling a formidable foe it may be more prudent and effective to first shape the battlefield than to simply add more soldiers.

In conclusion, countless anecdotal evidence and case studies suggest that the single most critical element for successful fire management in the WUI is collaboration with stakeholders from a diversity of worldviews (e.g., Dicus & Scott, 2006). Too often, fire managers in the U.S. have attempted projects only to fail due to unforeseen objections and resistance. Most commonly, fire managers believe that if only they could "educate" the public, the public would

willingly follow the direction of the fire manager. However, this attitude will generate fierce resistance and potentially doom a management proposal because it does not incorporate the worldviews and values of others who see the land from completely different perspective.

One of the most successful applications of a collaborative strategy in California has been the formation of local community FireSafe Councils, which purposefully seeks to include diverse interest groups from fields such as fire personnel, wildlife biologists, ranchers, developers, the insurance industry, the environmental community, builders, air pollution regulators, and others. While normally at odds with one another, these groups, through open dialog, consistently develop creative solutions that reduce wildland fire losses in the community while maximising other community values. To aid them, FireSafe Councils can apply for federal and state grants to fund educational products, fuels projects, pre-fire planning documents, and others. Whereas outside of this organisation many of the members have regularly been at odds with one another, FireSafe Councils allow them to see their collaborative ideas turned into action.

ACKNOWLEDGEMENTS

This project was funded by the California State University Agricultural Research Initiative. The author is grateful to Maurica Zimmerman for calculating community benefits with CITYgreen.

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INCLUDING SUPPRESSION EFFECTIVENESS IN FIRELINE GROWTH MODELS

R.O. WEBER, J.W. DOLD AND A. ZINOVIEV

Weber, R.O., Dold, J.W. & Zinoviev, A. 2008 06 25: Including Suppression Effectiveness In Fireline Growth Models. *Proceedings of the Royal Society of Queensland*, 115: 45-49. Brisbane. ISSN 0080-469X.

The inclusion of suppression effectiveness in fireline growth models is formulated as a system of differential equations. The model draws on earlier ideas using ellipses to model fire growth, particularly the head fire and flank fire rates of spread. It combines this with recent studies of the effect of fireline on spread rate and appends a single equation for the increase of the suppressed part of the fireline with time. Representative parameter values are used to illustrate this way of describing the effect of fire suppression activities on the fireline and to develop criteria for the likely outcome of containment activities.

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Wildland fire spread modelling has been an active research topic for many years and is the subject of a recent review by Pastor *et al.* (2003). Including chemical kinetics, such as in the paper by Assensio and Ferragut (2002), and the effects of suppression on fire growth and the eventual fire size; for example, Anderson (1989), are some of the options previously considered in order to make the models more realistic.

In the present paper, we introduce a new option for modelling fireline growth when there is active suppression applied from a set time after the fire was initiated. In particular, we use ideas from ellipse modelling of fire growth to provide the head fire and the flanking fire rates of spread. This is combined with information about the effect of fireline length on these spread rates (Cheney & Sullivan, 1997) and also linked with the rate of fireline growth in the presence of suppression. We use the mathematical framework of dynamical systems to illustrate this way of describing the effect of fire suppression activities on the fireline and to develop criteria for the likely outcome of containment activities.

It is anticipated that each component of the present model could be refined to account for more details of actual situations and that the result could then form a simulation module within fire incident management systems.

FIRELINE GROWTH MODEL

Let our fireline at any time t have a total length F(t). The requirement of a point ignition gives the initial condition that F(0) = 0. The total fireline should increase in length for all t > 0; at least until the fire is extinguished. We also formally divide the fireline into two parts: an active part L(t) and a suppressed part S(t) as illustrated in Figure 1.

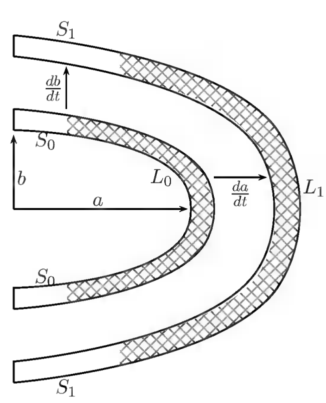

FIG. 1. Schematic of a Partly Suppressed Fireline

Inspired by the ellipse models (Anderson *et al.* 1982) and following on from Weber and Sidhu (2006), we write an equation for fireline growth as:

$$\frac{dL}{dt} = \alpha(\frac{da}{dt} + \frac{db}{dt}) - \frac{dS}{dt} \qquad (1)$$

Here α is a geometric constant with value between 1 and 2 (its precise value in any given fire will depend on the fire shape), $\frac{da}{dt}$ represents the head fire rate of spread, $\frac{db}{dt}$ represents the flanking fire rate of spread and $\frac{dS}{dt}$ accounts for the conversion of active fireline into suppressed fireline through suppression activities.

To be able to solve this model, we need several other pieces of information. From the literature; e.g. Cheney and Sullivan (1997), we deduce that the rate of forward spread depends upon the head fire width (for any given wind speed) in a way that can be described by the relation

$$\frac{da}{dt} = ROS_h(1 - e^{-\beta L}) \qquad (2)$$

where ROS_h is the potential rate of head fire spread under the prevailing conditions and β is a positive constant determined by fitting equation (2) to fire spread data.

We also assume that the flanking rate of spread, ROS_f, (typically a significantly smaller number than the head fire rate of spread) is described by a similar equation:

$$\frac{db}{dt} = ROS_f(1 - e^{-\beta L}) \qquad (3)$$

Note that the ratio ROS_h/ROS_f can be estimated from the length to breadth ratios often quoted for fires.
The rate of suppression is expressed in terms of available resources Q(*t*) through

$$\frac{dS}{dt} = Q(t) \qquad (4)$$

and this can be used to allow us to arrive at the final model

$$\frac{dL}{dt} = \alpha(ROS_h + ROS_f)(1 - e^{-\beta L}) - Q(t) \qquad (5)$$

subject to the initial condition $L(0) = L_0$ and which we must solve for L(*t*).

We note that it is not possible to write a closed form solution for L(*t*) for this model. Nevertheless it is reasonably simple to write a program to numerically determine the solution at any time, as we shall discuss later. First we shall discuss possible parameter values and examine some of the limiting cases where good approximate solutions can be found.

PARAMETER VALUES

To be able to use this model, we need to determine suitable values for each of the parameters. Clearly the values will depend upon the particular fires that might be considered, but for the purposes of illustration, we shall select as representative values ROS_h = 4,000*m/hr* and ROS_f = 1, 000*m/hr* and use these throughout this paper.

The parameter α will depend upon the geometry of the changing fireline and will probably vary from around $\alpha = 1.4$ for typical cases to $\alpha = 2$ as a likely maximum for extreme fire behaviour. This latter value can be used to obtain conservative estimates, as in the next section.

The time it takes the fireline to be sufficiently large to have reached its maximum potential spread rate determines the parameter β and examining the curves in Cheney and Sullivan (1997), we estimate $\beta = 0.03$ to be a typical value for moderate fires.

Values for the available suppression resources Q can be estimated for a variety of active suppression methods. A useful guide is McCarthy *et al.* (2003), from which we deduce that numbers such as 700*m/hr* are possible by a single dozer and significantly larger values are possible if several appliances and methods can be combined.

The initial fireline length L_0 can vary enormously and we will use several possible values in this paper to illustrate the potential application of the model.

LARGE FIRELINE LENGTH

For large fireline length, L(*t*), the head fire and flank fire rates of spread reach their full potential values. Then the non-linear exponential term becomes extremely small. If we also have constant available suppression resources (Q independent of time), then in this case the solution is very well approximated by

$$L(t) = L_0 + (\alpha(ROS_h + ROS_f) - Q)t \qquad (6)$$

For the suppression activities to be successful in this limit, we require

$$Q > \alpha(ROS_h + ROS_f) \qquad (7)$$

and we can estimate the suppression time required by

$$t_{sup} = L_0/(Q - \alpha(ROS_h + ROS_f)) \qquad (8)$$

To illustrate this, we use the values $ROS_h = 4,000$*m/hr* and $ROS_f = 1{,}000$*m/hr* as discussed in the previous section, we set α = 2 to be representative of extreme conditions, we assume quite active suppression such as Q = 12,000*m/hr* and we begin with an initial fireline of length $L_0 = 4,000$*m*.

As the available suppression resources satisfies equation (7), we can be confident that in this case the suppression activities will be successful and we can estimate from the equation for tsup that it will take approximately two hours to contain this fire.

Note that the time scales linearly, so that if the initial fireline is twice as long, then the suppression time is also twice as long.

VERY SMALL FIRELINE LENGTH

For very small fireline length, the head fire and flank fire rates of spread are very small, so that suppression is the dominating effect and the approximate solution to our model (again assuming constant Q) in this case is

$$L(t) = L_0 - Qt \qquad (9)$$

reflecting the well known fact that every fire can be extinguished provided suppression arrives sufficiently early. From this we can also obtain an estimate of the "early extinction time", t_{ee} as

$$t_{ee} = L_0/Q \qquad (10)$$

For example,
if $L_0 = 10$*m* and $Q = 20$*m/hr*, then $t_{ee} = 30$ minutes.

GENERAL CRITERIA AND ESTIMATES

In general, we are interested in the application of the model to situations where there is a dynamical interaction between the fire growth and the fire suppression activities and we are particularly interested in using the model to obtain criteria for the likely success and estimates of the time it will take for successful suppression.

To this end, in this section we analyse the model with as few assumptions as possible. We begin by noting that the exponential $1 - e^{-\beta L}$ is always less than unity for any fireline length L. Hence, provided that suppression activities are maintained so that

$$\frac{dS}{dt} > \alpha(ROS_h + ROS_f) \qquad (11)$$

fire suppression will always succeed no matter how large the initial fireline length is at the commencement of suppression activities. This is really just a common sense conclusion and as a conservative estimate to guarantee success we would also let α = 2 and then restate our criteria in words as

- calculate potential head fire and flank fire rates of spread
- add these and multiply by two
- maintain the rate of construction of suppressed fireline so that is greater than this last number

An example calculation illustrating this is as already done in section 5 and we note that this shows that treating every fireline as a potentially extreme fire will provide conservative estimates for the resources required for successful suppression.

While this guarantees success, there is still the possibility of suppression success with less suppression activity. For example, we can maintain decreasing L (i.e. keep $\frac{dL}{dt} < 0$) at all times provided
If we apply this to the initial fireline length L0, assume

$$\alpha(ROS_h + ROS_f)(1 - e^{-\beta L}) < Q(t) \qquad (12)$$

constant Q and rearrange, we obtain
as an estimate for the maximum initial fireline length

$$L_0 < -\frac{1}{\beta}\ln\left(1 - \frac{Q}{\alpha(ROS_h + ROS_f)}\right) \qquad (13)$$

which can be suppressed with resources Q. Note that this estimate can only be applied provided that the inequality $Q < \alpha(ROS_h + ROS_f)$ is satisfied. This means that this estimate must be applied with care. An example is the parameter values used in section 5 but with the significantly smaller value for the suppression resources: Q = 9,000*m/hr*. Then we obtain a maximum initial fireline length of only 76*m* (rounded down) for suppression to be successful.

We could also estimate the time it will take to suppress any such initial fireline with L_0 less than this with the equation

$$t = \frac{L_0}{Q - \alpha(ROS_h + ROS_f)(1 - e^{-\beta L_0})} \tag{14}$$

which gives a rather small time of $0.04hr$. In fact, these approximations are not uniformly valid as L decreases from a value like 70 to zero and it is typically much easier, safer and accurate to simply solve the differential equation numerically as will be done in the next section.

Contrasting these findings shows that, for this example, suppression resources of 12,000*m/hr* would be able to succeed against any initial fireline, whereas suppression resources of 9,000*m/hr* would only be able to succeed provided that the initial fireline length is less than the rather modest value of 76*m* (note that numerical solution techniques as discussed in the next section will revise this value to the more accurate 69*m* which indicates that there is a small but potentially important level of error with the approximations that result in equations 13 and 14).

NUMERICAL SOLUTION

The model presented in this paper can be easily solved numerically using a variety of routines and software applications. Possibly the simplest is the Euler method where the derivatives are approximated by differences and used to update the solution in small time steps. Provided the time steps are small enough the error in this method is also kept reasonably small, the results are obtained very quickly and the method can be easily programmed with any software such as fortran, basic, matlab or excel.

The essence of the method applied to our model can be stated as follows. Begin with an initial fireline length L_0 and update it in small time steps Δt according to

$$L_{new} = L_{old} + (\alpha(ROS_h + ROS_f)(1 - e^{-\beta L_{old}}) - Q(t))\Delta t \tag{15}$$

We also note that this numerical method easily allows us to calculate the solution even when the suppression resources, *Q(t)*, vary with time; an obvious improvement on our previous analytical approximations.

To illustrate the numerical solution of the full model, we select the values for the model: $\alpha = 1.4$, $\beta = 0.03$, $ROS_h = 4{,}000m/hr$, $ROS_f = 1{,}000m/hr$ and the rather modest initial fireline length of $L_0 = 1{,}000m$. Then we can immediately see the consequences of varying the available suppression resources Q. If Q is less than 7,000*m/hr*, the fire will continue to grow and the suppression activity will never be able to contain the fire.

CONCLUSION

To recap the essential features of our model and the solutions we wish to highlight the most salient results in the present paper.

To establish a safe, conservative estimate for successful suppression, we can refer to equation 7 which allows us to calculate a level of suppression activity which will ensure success. Then we can also estimate the time for which this level of suppression needs to be maintained for any initial fireline length using equation 8. It should be emphasised that this is most likely to be the usual situation and that equations 7 and 8, with suitable field testing, are most likely to be the most useful simple results from the present model.

On the other hand, if we wish to adopt a strategy of early intervention, then (as long as the initial fireline length is sufficiently small as estimated by $L_0 < \frac{1}{\beta}$), equation 10 can be applied to determine the time required for suppression.

The numerical solution presented here provides a way of simulating the growth of the fireline with any desired suppression strategy. With suitable field verification and calibration this could be used to predict the likely failure or success depending upon the level of applied suppression resources and the size of the fireline when the suppression activities first commence.

ACKNOWLEDGEMENTS

The authors would like to thank Ross Bradstock, Geoff Cary, Karen King, Malcolm Gill and Greg McCarthy for helpful discussions during the course of this work and to acknowledge the Bushfire CRC, the Leverhulme Foundation, the UNSW at ADFA and the University of Manchester for financial support for this work.

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BUSHFIRE MANAGEMENT: WHERE, WHY AND HOW ECONOMICS MATTERS

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Ganewatta, G. & Handmer, J. 2008 06 25: Bushfire Management: Where, Why And How Economics Matters. *Proceedings of the Royal Society of Queensland*, 115: 51-57. Brisbane. ISSN 0080-469X.

Bushfires destroy existing resources while fire prevention and suppression require resources that have alternative uses. Consequently, the threat of bushfires alters resource allocation affecting the well being (welfare status in the language of economics) of society. This paper explores the potential of using an economic framework to improve bushfire management through better use of scarce resources. The effect of bushfires on the resource base and associated socio-economic impacts, resource allocation for fire management, and public policy issues are some areas of interest for economic analysts. The uncertainty of the potential damage from unmanaged fire events and the impact of infrequent and massive fire events on regional economies are some of the challenges for analysts attempting to introduce economic thinking into fire management decision-making. Economics provides a standard framework for valuing resources damaged by bushfire and human suffering. Impact assessment of major bushfire events on regional economies, and resource allocation for different programs are also areas where economic methodologies can help by better allocation of scarce resources.

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Bushfires remain an inevitable natural event in many parts of the Australian landscape due to climate, the nature of the ecosystems and the existence of many ignition sources (McCormick, 2002; Dovers *et al.*, 2004; McGee & Russell, 2004). Frequent and prolonged drought and ever increasing human interaction with the natural environment has increased the threat of bushfire throughout the country. Bushfires produce distinct impacts on the resource base of society, affecting economic behaviour. This paper explores the potential of economics to improve bushfire management. Economic analysis not only provides a framework for analysing the behaviour of society, but also provides guidelines for the best use of scarce resources.

BUSHFIRE AND ECONOMICS

Bushfires destroy available resources for production and consumption and demand scarce resources be allocated for fire management. Consequently, society needs to have ways of allocating resources for the best social outcome. We present here a discussion on important issues associated with bushfires in relation to the concerns of economic analysis.

BUSHFIRE AND THE RESOURCE BASE OF SOCIETY

The damage incurred by the sudden onset of any destructive event primarily includes productive capital such as infrastructure and may effectively destroy the means of production as well as stocks (Pelling *et al.*, 2002). Bushfires interfere with productive capital stocks, natural resources and environmental services affecting production and consumption possibilities. Replacing or repairing affected capital stocks requires resources that would have otherwise been allocated for other productive purposes or consumption. The average annual damage cost from disastrous bushfires in Australia is estimated at $77 million (BTE, 2001). This figure does not include the financial losses caused by a large number of small-scale bushfire events every year. In addition to the direct financial costs of damage to assets and productive resources, intangible damage to environmental resources and services such as water yield and quality are also an important part of the economic losses from bushfires. During the period from 1967 to 1999 bushfire accounted for 39 per cent of the fatalities and 57 per cent of the injuries associated

with natural disasters in Australia (BTE, 2001).

Reducing the risk of bushfire requires a considerable amount of financial and non-financial resources that could otherwise be made available for alternative uses. For example, for the financial year 2003/04 in the State of Victoria, Australia, fire management programs carried out by the Country Fire Authority (CFA) had an operational budget of $186.5 million. The CFA employs 1100 paid staff while drawing on the services of 58, 000 volunteer fire fighters to meet its human resources requirement (CFA, 2005). Department of Sustainability and Environment (DSE) of Victoria spent $34 million on fire management programs for the financial year 2002/03 (DSE, 2003). During the 2002-03 bushfire season, state and territory governments spent more than $251 million on fire suppression activities and deployed over 140 fire suppression aircraft, costing over $110 million alone (Ellis *et al.*, 2004).

With the expansion of cities and an increasing desire to live close to nature, there has been a considerable enhancement in the threat of bushfires to human life. An improved understanding and appreciation of the importance of environmental resources and services has also increased the community expectations in bushfire suppression. Thus, investment in prevention programs and the acquisition of effective suppression capabilities are important requirements that have resulted in greater demand for resources.

ISSUES ASSOCIATED WITH BUSHFIRE MANAGEMENT

It is often impossible for larger communities to organise the bushfire protection required, and market forces are not the best way of providing this service. Instead, the provision of bushfire protection often falls into government hands, for which funding is largely raised through taxes and public levies. Thus the provision of bushfire control and the level of appropriate protection are decided through the political system. The process of political decision-making is subject to the influence of pressure groups and could result in government failure to provide efficient levels of fire management services required by society or individual communities.

The risk of damage to property and life is high in isolated and remote communities. These communities may not be able to acquire adequate equipment with the limited resources available. However, such large investments may be necessary for adequate fire protection. The provision of aerial fire suppression units is one example that required federal level resource allocation due to very high capital outlay for the initial investment. Thus the government faces the issue of providing adequate protection for everyone while choosing a suitable approach for financing the fire management services. These areas are of interest for policy makers as inherent characteristics of fire management services could yield inefficient resource uses.

The resources allocated for fire suppression should have alternative uses that benefit the majority of society rather than smaller communities or wealthy individuals. Thus the decision to provide publicly funded fire suppression services for private property protection is an important concern since such government intervention may induce private individuals to ignore the risk associated with bushfire in their decision making process.

Fuel reduction burning to protect remote communities can result poor air quality and visibility from smoke and haze affecting urban populations (Loomis *et al.*, 2004; Loureiro *et al.*, 2004). The elderly and asthmatics are more likely to suffer from ill health and associated problems due to the poor air quality. Fuel reduction burning also runs the risk of escaping from control and causing damaging bushfires. Fire that escapes from fuel reduction burnings can produce unexpected fire events resulting in huge losses. Public and political opposition to these management practices, as a result of these deleterious effects, leads to a build up of fuels, thus increasing the risk of catastrophic fire (McCormick, 2002). The decisions to adopt such activities are an important concern of public policy making process.

SOCIAL AND ECONOMIC IMPACTS OF BUSHFIRE

Bushfire events can produce serious indirect effects on regional economies in addition to direct economic losses. For example, the social and economic cost of the first six months after the 2003 bushfires across Gippsland and the North East Region of Victoria (in terms of loss of income and production) was estimated at $121 million (Gangemi *et al.*, 2003). Areas that depend highly on the tourism industry could have been affected from the disruption of tourist activities resulting in long-term consequences. On the other

hand, there may be locally positive impacts such as business expansion due to increase in timber recovery or construction financed by insurance payouts and government compensation.

Taking tourism as an example, loss of employment due to diverted tourist flows towards other attractions can be a gain for another region resulting in economic benefits. These changes are transfers from one region of the economy to another from a national economic perspective. However, such changes can make a big difference to the lives of the people in the region of concern. Those who are unemployed may not be able to move freely to other areas due to associated socio-economic factors. In many cases, authorities may have to divert resources for recovery programs. For example, the Victorian government allocated $86 million to support recovery programs in the Gippsland region after the 2003 fire season (Whittaker & Mercer, 2004). Thus the economic and social impacts of bushfire are of important concern for policy-makers which, in turn will affect resource allocation.

ECONOMIC STUDIES OF BUSHFIRE MANAGEMENT IN AUSTRALIA

Researchers such as Luke & McArthur (1978); Hatch & Jarret (1985) and Mulers (1985), have attempted to examine the economic aspects of bushfire management in Australia. Loane & Gould (1986) analysed the costs and benefits of introducing aerial fire suppression capabilities which provided evidence for public debate on the issue. The costs and benefits of using aerial suppression were estimated and results indicate a positive saving for several types of large and small aircrafts and helicopters under certain conditions. Benneton *et al.* (1998) presented an economic evaluation of the Fire Management Program (FMP) of the Victorian Department of Natural Resources and Environment, which is responsible for the prevention and suppression of fires on public land in the state of the Victoria. The analysis estimated probable damage from fire incidences, by representative year with and without the presence of public FMP in Victoria, using a fire simulation model. The results of the cost benefit analysis show the ratio of benefits to costs in an average year to be approximately 24 to 1, indicating that every dollar of public resources allocated brings 24 dollars worth of benefit in terms of assets not destroyed by bushfire. The use of the fire simulation model to generate information on the probable damage under alternative scenarios is an important aspect of the study.

The Australian Bureau of Transport and Regional Economics (BTE) has examined the economic cost of disaster level bushfire events in Australia. It shows that Australia experiences disaster type bushfire events frequently and bushfire is the most dangerous natural hazard in terms of risk to human life (BTE, 2001). The study presents an approach to estimating the economic cost of disasters and bushfire events. The report firstly identifies the difference between the financial and economic analysis, and secondly separates the direct and indirect costs, as well as tangible and intangible costs of disasters. However, estimating the economic effect of a bushfire event does not identify the beneficial effect of fire events. The methodologies developed are limited to the assessment of the economic loss caused by disaster level events. Nevertheless, most of the methodological approaches suggested can be used in valuing the economic impact of a bushfire incident.

CHALLENGES IN THE USE OF ECONOMIC ANALYSIS IN BUSHFIRE MANAGEMENT

Bushfires produce large impacts on the environment, creating intangible costs and benefits to society. The social benefits and costs of the intangible impacts of bushfire are not expressed in monetary value and are therefore difficult to introduce into economic analysis. The use of economic analysis in the presence of a larger magnitude of intangible benefits and costs only produces incomplete information. Decisions made on the basis of incomplete information may not be socially and politically acceptable. Incomplete information on the costs and benefits of bushfire limits the use of economic analysis for fire management decisions based on economic efficiency criteria.

In the economic way of thinking, a cost avoided from an action is a benefit of that action. The economic benefits of fire suppression derive from the damage averted from suppression activities (Handmer *et al.*, 2002; de Mendonca *et al.*, 2004). However, damage averted from fire suppression is impossible to identify under normal circumstances. The spread of fire depends on a number of different variables such as climate, environment, topography, fuel build up, etc (i.e. Li & Magill, 2001). Thus, potential damage from any fire can only be elicited using a fire simulation modelling approach. Instead, the actual damage from fire is visible and easy to assess. Thus, the use of economic efficiency measures such as cost benefit analysis in decision making remains a difficult task in fire management.

Suppression of a small fire may require much larger resources than the value of the protected resources. Such action is not economically justified. However, any small bushfire could develop into a disastrous fire bringing devastating economic losses if unattended at an early stage. Bushfire events could also result in the economic decline of regional communities due to the change of economic activities in that area. Thus the viability of economic analysis itself on decision-making may not be applicable on all occasions.

ECONOMICS FOR BUSHFIRE MANAGEMENT

It is necessary to incorporate economic information in fire management decisions as bushfire interacts with the resource base and produces serious economic impacts on affected communities. Nevertheless, operational difficulties limit the use of economics in bushfire management. Identification of appropriate analytical frameworks and limits to their use are therefore useful for the efficient allocation of scarce resources in bushfire management.

VALUING THE RESOURCES AFFECTED FROM BUSHFIRE

Social decision-making on bushfire management should be based on economic costs and benefits of bushfire rather than the financial cost for individuals. The economic costs of bushfire include the opportunity cost of resources used in fire management and the social value of resources affected. Thus, it is required to estimate the economic cost and benefits of bushfire using standard economic frameworks focusing on the real value of resources.

(1) Valuing the resources used for fire suppression

Resources used in bushfire management programs include capital and human resources. Resources that are used for public expenditure could be used to produce other goods and services instead, that is the opportunity cost of public expenditure. The direct budgetary outlay for fire management and associated services could be considered as an identical measure of the opportunity cost of the resources allocated. With the presence of an efficient market (where the use of resources does not affect the market equilibrium) then the financial cost of resources used can be considered as the opportunity cost of the resources forgone (Boadman *et al.*, 2001). Volunteers are highly involved in the human resources associated with fire management programs. However, where the services of volunteers are not available for fire management agencies, remuneration of paid workers can be considered as the cost of the human resources involved with fire management activities.

(2) Valuing the damage to the assets and outputs

The damage from bushfire to assets, results in loss of return over the lifetime of that asset. The economic cost of damages to a capital asset is the sum of the discounted present values of the flow of economic returns that would have been produced by the asset. Estimation of the value of damage to houses requires further attention. The estimation of the value of a damaged house from real estate prices for the region could not be justified as house prices are highly influenced by land price. The use of a replacement price or the cost of building a similar house can also be questioned, as the level of satisfaction generated from the new house must be higher than that of the used house due to the depreciation, while the opposite may be the case for others. Thus, both of these measures are not acceptable as a reasonable measure of the value of a lost house. Alternatively, net present value of the expected rental income over the life of the house could be used as an appropriate measure of the value of the damaged house.

Bushfires destroy timber resources, crops, pastures and livestock products. The economic value of the damaged products should be valued using the shadow prices that represent the real value of the outputs concerned. The damage to timber depends on the intensity of the fire and the maturity of the trees and thus there will be a salvage value of timber. The economic loss from the damage to the timber is equal to the net present value of the expected income, less the salvage value of the resources. The value of damages to crops, pasture and livestock products also requires quantification using shadow prices that reflect the actual resource cost of the output.

(3) Valuing environmental costs and benefits of fire

The natural environment not only provides sources of material inputs for the economic system but also life support services for people in the form of a breathable atmosphere, a liveable climatic regime, a variety of amenity services and acts as a waste receptor service (Freeman, 2003). Bushfires interact with the production of these wildland services and affect their use and non-use value. On the other hand, bushfires produce some positive effects on the environment that have economic values for society.

Valuation of costs and the benefits of environmental

impacts require the use of non-market valuation approaches, most widely favoured by Environmental Economists. The valuation of environmental damage is useful to obtain a more accurate cost benefit analysis of alternative policies, to demonstrate the relative importance of environmental consequences and to make environmental damages more palpable (Glover & Jessup, 1999). Since non-market valuation studies require time, resources and expertise, appropriate use of information generated in existing studies may be used for estimating the costs and benefits of individual events.

(4) Valuing the effect on human life

Estimates of the Value of Statistical Life and cost of injuries will be useful in evaluating or comparing alternative programs proposed to reduce risk of death and injuries from bushfire. Thus, the use of value assigned for human life from other studies could be an acceptable approach in assessing the impact of a fire management program. There are also techniques developed for valuing human injuries through the cost of treatment and lost income for the affected period. Such techniques may be used to estimate the cost of injuries, though they are lower bound estimates as there is no consideration given for the suffering from injuries.

ECONOMIC IMPACT ASSESSMENT

Economic impact assessments evaluate the regional effects of actions on prices, outputs, employment and other economic factors, focusing on how those effects are distributed across the region. The assessment of a bushfire impact on the regional economy provides an insight into the consequences or potential consequences of a bushfire event in the area. In the assessment, the use of economic frameworks would provide a common basis for valuing the impact, giving wider acceptance by policy makers, politicians and communities. Such information would help decision makers to design recovery programs after bushfire disasters in regional communities (Handmer *et al.*, 2002). Information on the possible economic impacts of bushfire on regional economies would also be useful for political decision making in order to justify new mitigation strategies for remote communities.

ALLOCATION OF RESOURCES TO FIRE MANAGEMENT PROGRAMS

Economic methodology can provide a conceptual framework for decision-makers to allocate resources for specific management programs to achieve socially desired objectives.

(1) Optimum resource allocation

The objective of bushfire management is to minimise the total resource loss from damage and suppression efforts. Enhanced preparedness for bushfire events helps to reduce the damage from the destruction that comes at an extra cost for society. Economic frameworks can be adopted to identify the optimal resource allocation for fire management that minimises the net resource outlay from society.

Russell (1970) presents a graphical model to identify the best level of adjustment through the identification of minimum sum of the adjustment cost and expected losses from damage. Similar methods known as the least cost plus loss (C + L) approach and cost plus net value change (C + NVC) exist for forest fire management to identify the lowest level of resources required for fire management programs. Both approaches that attempt to identify the level of resources to be allocated for bushfire management programs could be adopted for resources allocation on bushfire management.

(2) Provision of additional resource to fire management programs

Fire management programs require additional resources for expanding their capacities to meet the challenges ahead. Policy makers then face the question of the magnitude of the expected return or net saving of resources from the unit investment as a means of justifying the intended investment. Analytical tools such as Cost Benefit Analysis (CBA), used in economics would be an appropriate methodology for answering the concerns of policy makers.

(3) Identifying alternative approaches for bushfire suppression

Fire managers can use economic tools to make informed choices from alternative fire suppression technologies. Fire managers would also be able to optimise resource use, allowing them to make the most of the available resources. CBA framework or Cost plus Loss approaches are important analytical tools that could be used to assist fire managers in making decisions on the use of alternative fire suppression strategies.

CONCLUDING REMARKS

The occurrence of bushfires produces distinct impacts on resources, affecting the economic behaviour of society. Bushfires destroy existing resources, while fire management programs require the allocation of resources that may have had alternative uses.

Consequently, the threat of bushfire affects the economic well being of society by altering resource allocation. This paper explored the potential of using economic frameworks to improve bushfire management by making better use of scarce resources.

The effect of bushfires on the resource base of society, the associated socio-economic impacts, and resource allocation for fire management, are areas of concern for economists. This paper identifies the challenges of using a conventional economic framework to enhance fire management decision-making. The benefits of fire suppression derive from the value of the damage averted and it remains difficult to estimate the extent of the potential damage. The introduction of infrequent and massive fire events and their impacts on regional economies in the fire management decision-making process remains a challenge for the analyst.

In spite of these challenges, the use of an economic approach is acceptable from a social decision making perspective, as it considers the net effect on society rather than on individuals or groups of individuals. This paper shows a number of areas where economics can contribute to fire management decision-making. Economics can provide a standard framework for valuing human suffering and resources affected by bushfires, especially the more intangible effects, to reflect their true social values. Economic methodologies can also help with better allocation of scarce resources in fire management, through the assessment of major bushfire events on regional economies and resource allocation for different fire management programs.

ACKNOWLEDGEMENTS

This paper is based on the work funded by the Bushfire Cooperative Research Centre (CRC) and RMIT University. Their support is gratefully acknowledged, as are the valuable comments and editorial support provided by Thomas Lowe and Wei Choong of the Centre for Risk and Community Safety of RMIT University.

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THE ECOLOGY OF FIRE – DEVELOPMENTS SINCE 1995 AND OUTSTANDING QUESTIONS

R.J. WHELAN

Whelan, R.J. 2008 06 25: The Ecology Of Fire – Developments Since 1995 And Outstanding Questions. *Proceedings of the Royal Society of Queensland*, 115: 59-68. Brisbane. ISSN 0080-469X.

A great deal is already known about fire ecology in Australia, because careful observation of fire effects have been informing fire management for many thousands of years and scientific study of fire ecology has been going on for over a century, especially in the fields of forestry, evolutionary ecology, and land management. In this paper, I review some of the key questions of fire ecology identified in *The Ecology of Fire* (1995) for which I perceive there is a need for an expanded research effort and for better communication to politicians, policy makers, land managers, and the public at large. These include (i) better knowledge of fire history in particular areas, (ii) a more sophisticated understanding of what is meant by 'fire mosaic' and how different spatial patterns of fire might affect ecological processes, (iii) developing tools for predicting ecological responses to particular fire regimes, and (iv) the more comprehensive use of experimental and adaptive management at a landscape scale, given that environmental conditions will always be changing and ecological knowledge will never be complete.

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Bushfire is on the agenda more than ever… internationally. The attention of the public and politicians has been captured by extensive media coverage on big fires in the last 5 years: Portugal, France, California, Colorado, South Africa, Indonesia, the Amazon – and 2001-02 and 2002-03 in south-eastern Australia. The various enquiries that have followed these fire events, at least in Australia (e.g. the NSW Joint Select Committee on Bushfires 2002, the Victorian Government's Inquiry into the 2002-2003 Victorian Bushfires (Esplin *et al.*, 2003), the House of Representatives Select Committee Inquiry into the Recent Australian Bushfires (Nairn, 2003), and the Council of Australian Governments National Inquiry into Bushfire Mitigation and Management (Ellis *et al.*, 2004), have revealed many misconceptions about fire characteristics and about the ecological impacts of bushfires. For example, the Hansard record of submissions to the House of Representatives Select Committee Inquiry includes the following:

> *Wouldn't that mosaic type burning allow animals to move into another area and not be burned out, whereas a feral fire would burn out the whole area and, as we saw in many parts of Australia last summer, there would be gullies full of dead native animals? … I call them 'feral' because of their impact – the intense feral fires that burn asphalt.*

Ms S. Panopoulos, House of Representatives Select Committee on the Recent Australian Bushfires. Hansard 8th July 2003, p. 40.

> *…a lightning strike in there would destroy an enormous amount of biodiversity, which has now happened. It has destroyed the biodiversity to the point, as I said earlier, where it has vapourised any known seed stock that may have been below the ground, because it sterilised the earth to 40 feet below the surface in some areas.*

Mr A. Schultz, House of Representatives Select Committee on the Recent Australian Bushfires. Hansard 8th July 2003, p. 41-42.

It would be well worth assessing the issues raised in these public airings of people's perceptions, because the nature of the misunderstandings may point to ways of better educating the Australian community about ecological effects of fire. The various inquiries have also highlighted the demands that the development of policy in relation to fire management and mitigation will increasingly make on ecology, and have revealed significant gaps in our knowledge. For the purposes of this paper, I focus especially on the challenge of achieving life and property protection without compromising biodiversity conservation. These dual responsibilities of many land managers are often in conflict and in some situations there may not be satisfactory compromises – the situation highlighted

in a My Fair Lady song: "…make a plan and you will find, that she has something else in mind, and so rather than do either you do something else that neither likes at all!"

We already know a great deal about fire ecology, because careful observation and experimentation have been informing indigenous management of fire, to achieve specific management objectives, for many thousands of years (e.g. Hill, 2003; Liddle, 2003). Scientific study of fire ecology in Australia has been going on for many years too, especially in the fields of forestry, evolutionary ecology, and land management for conservation. For this paper, I was asked to review developments in fire ecology since *The Ecology of Fire* (Whelan, 1995), a task that is too large for this article. Much of the recent published work is summarised in a number of excellent recent monographs and the references therein (Table 1), especially *Flammable Australia* (Bradstock *et al.,* 2002), which reviews the state of knowledge on fire and biodiversity for a range of different ecosystems. I focus here on some key areas in Australian fire ecology in which I perceive a need for a renewed or broadened research effort, particularly in relation to land management

In writing *The Ecology of Fire,* I identified a set of questions, in each of these main topic areas, which I saw to be particularly important yet had been ignored or poorly studied. These are summarised in Table 2. The studies presented at the Bushfire 2006 conference, some of which are presented in this volume, present an interesting test of the development of fire ecology in recent years, especially in relation to their coverage of ecological processes, taxa and approaches used. In the following sections, I have selected some important areas in which land management for ecologically sustainable bushfire mitigation and management make demands on ecological knowledge, and I explore the limits to our current ability to satisfy these demands.

FIRE HISTORIES

The ecological and evolutionary forces moulding the characteristics and distributions of species in fire-prone landscapes could be more thoroughly explored if we had information about fire histories at a range of scales. I came to the conclusion in 1995 that better fire histories are needed, with more techniques in more communities (Table 2). This is still the case, though there have been significant developments. The

TABLE 1. Recent monographs addressing current knowledge in fire ecology

Abbott, I. & Burrows, N. (eds) (2003) "Fire in the Ecosystems of South-west Western Australia: Impacts and Management", Backhuys, Leiden.
Andersen, A.N., Cook, G.D. & Williams, R.J. (eds) (2003) "Fire in Tropical Savannas: The Kapalga Experiment. Springer, N.Y.
Bradstock, R.A., Williams, J. & Gill, A.M. (eds) (2002) "Flammable Australia: The Fire Regimes and Biodiversity of a Continent", Cambridge University Press, Cambridge.
Bowman, D.M.J.S. (2000) "Australian Rainforests: Islands of Green in a Land of Fire", Cambridge University Press, Cambridge
Cary, G., Lindenmayer, D. & Dovers, S. (eds) (2003) "Australia Burning: Fire Ecology, Policy and Management issues", CSIRO Publishing, Melbourne.
Esplin, B., Gill, A.M. & Enright, N. (2003) "Report of the Inquiry into the 2002–2003 Victorian Bushfires", State Government of Victoria, Melbourne.
Ellis, S., Kanowski, P. & Whelan, R.J. (2004) "Council of Australian Governments – National Inquiry into Bushfire Mitigation and Management", Australian Government, Canberra.
Mackey, B., Lindenmayer, D., Gill, A.M., McCarthy, M. & Lindesay, J. (2002) "Wildlife, Fire and Future Climate", CSIRO Publishing, Melbourne.
NSW Nature Conservation Council Conference Proceedings 1998, 2000, 2002, 2004 (http://www.nccnsw.org.au)

Table 2. 'Outstanding questions' identified in Whelan (1995)

Chapter	Issues and Questions
Fire the Phenomenon	Better fire histories needed, with more techniques in more communities. How much do extremes in inter-fire intervals vary from the average fire period? What are the effects of topography and local climate on fire patchiness? To what extent are unburned patches consistent in successive fires? Simple, repeatable estimation of fire characteristics, of ecological relevance, are needed. We need more information on post-fire physical conditions.
Survival of Individual Organisms	We are lacking knowledge of the effects of season and frequency of fires on mortality of resprouting woody plants. More research is needed on the dynamics of soil- and canopy-stored seed banks. What conditions of fire and environment favour the evolution of bradyspory (serotiny)? Why is there growth-stimulation in woody plants after some fires but not others? What are seed dispersal distances in relation to spatial patterns of fires? How does life-history influence survival of fire by animals? How does this interact with the season of burning and fire characteristics? What are the responses to fire in historically fire-free environments?
Approaches to Fire Studies	"No one would now dream of testing the response to a treatment by comparing two plots, one treated and the other untreated" (Fisher and Wishart, 1930 – in Underwood, 1986). The design of a study must be related to the question – which defines the inference(s) that will be made from the results.
Plant Populations	How does fire patchiness affect the proportion of plants that survive? How does patchiness or extent influence post-fire herbivore-plant interactions? How does pre-fire seed dispersal affect survival of the seed bank? How does post-fire seed dispersal determine seed survival to germination? Do causes of seedling mortality vary among seasons? How do plant populations respond to a sequence of fires? Do the chance elements of post-fire climate have an over-riding effect on plant population dynamics?
Animal Populations	How do different sorts of fires affect mortality, emigration and survival? What is the importance of recolonisation *vs.* survival within a burned area? Are animals found in refuges after fire those that happened to be there prior to the fire or did they actively seek out refuges? What is the relative importance of food, cover and predation in post-fire population dynamics? What explains highly variable results of post-fire populations of soil and litter invertebrates?
Communities	We badly need experimental studies of changes in community parameters with replication of fires. We particularly need experiments manipulating fire frequency and season over long time spans. More than a single trophic level needs to be included in experimental studies. More focus on the role of below-ground interactions (e.g. mycorrhizae). A critical review of plant succession theory as it relates to fire ecology in different ecosystems is overdue. How important are specific conditions in community changes after fire (e.g. post-fire climate, pre-fire community composition)?
Management	"It is obvious that there is unlikely to be sufficient ecological information to be certain of the ecological effects of *any* prescribed fire regime. Hence, management will have to be experimental." "It is unlikely that all objectives for land in multiple use will be able to be achieved under one fire regime"

summary by Gill (2002) of the range of sources of evidence for past fire regimes in SW Australian forests is applicable to the inference of fire history in general. The techniques he reviewed include:

- interpretation of burning practices of indigenous people;
- monitoring and historic records;
- 'annual' rings and fire scars;
- banding in leaf-bases of *Xanthorrhoea*;
- demographic structure of plant populations;
- inference or modelling based on plant life histories;
- palynological and charcoal data.

Some of these techniques are contentious (see Enright *et al.,* 2005) and some are applicable in only a limited number of situations. Some provide point-based and others area-based estimates of between-fire intervals; a distinction that is very important.

While the research challenges of inferring past fire regimes are important and fascinating, high-quality monitoring is needed today to inform the decision-makers of the future. Satellite-based mapping of fire–affected areas exists at different scales for various parts of Australia and is widely available, from a range of sources, via the internet. The COAG Bushfire Inquiry (Ellis *et al.,* 2004) considered that this is such an important development that it recommended: *That the Australian Government and the state and territory governments jointly provide additional resources and work in partnership to establish and refine a national program of fire regime mapping.* In this conference, the paper by Barrett (2006) on the use of satellite imagery to model bushfire severity in the 2003 NSW/ACT fires shows that we have come a long way since 1995, and approaches like this will allow future assessment of how factors such as tree mortality, recruitment, erosion, and community composition vary in relation to fire intensity after a particular fire event. A particular challenge for satellite mapping is improving the detection of fires under cloudy conditions and of fire severity in areas with dense forest canopies. Barrett (2006) also reminds us of an important feature of bushfires in heterogeneous landscapes – namely, that they are not uniform within the fire boundaries.

What has changed since 1995? Although we do not have precise fire histories for ecosystems in most parts of the continent, and some results are still contentious, it is now clear and generally accepted that pre-Aboriginal and pre-European fire regimes varied from one place to another, at various scales, strongly influenced by climate and ignition interacting with landscape and vegetation (see Kershaw *et al.,* 2002). We recognise that these differences in fire history among regions will have shaped the evolution of organisms. It is also clear that European settlement has resulted in a marked change in fire regime in many areas, although once again there are few empirical data that would allow precise quantification of the change. Nevertheless, as ecologists we recognise that the changes in fire regime that have accompanied European settlement, population growth, forestry and urban expansion are certain to have different effects on organisms, depending on their evolutionary histories. Additional effects will certainly accompany the future changes in fire regime caused by climate change, ever increasing landscape fragmentation, and alteration to plant communities by weed invasion. We have not yet communicated this level of understanding to the general public.

MOSAICS OF FIRE AGES VS FIRE REGIMES

Scientific studies in many regions suggest that the continuous application of a single fire regime over a landscape may be detrimental to biodiversity (see, for example, a range of studies presented in Abbott & Burrows, 2003 and Andersen *et al.,* 2003). The corollary, that biodiversity would best be protected with a fire "mosaic" in the landscape, has been seized on as a solution to the trade-off between biodiversity conservation and protection of lives and property, and has been presented as such to recent bushfire inquiries, as a fuel-reduction prescription. It is important to define the term "mosaic" here, because it is being used undefined, both in the scientific literature and in policy statements, in two different ways. One is to describe a landscape that has patches of vegetation of different ages after fire, even though each patch might be being burnt with the same return time. This is not a mosaic of fire regimes; it is a mosaic of fire ages. Such a prescription may protect adjacent properties if the return-time were short enough, but it would not sustain a species of animal, for example, that is fire-sensitive and dependent on dense cover in the ground and mid-storey layers. On the other hand, a landscape with a mosaic of fire regimes would have some patches that are rarely burned, some more frequently, some in each season, some small, some large, some high intensity, and some cooler.

Creating a mosaic of fire regimes across a landscape, with fire intervals, seasons and intensities in the

mosaic that are appropriate for particular ecosystems, appears to be a reasonable goal for ecological burning in a range of ecosystems (e.g. spinifex grasslands in arid Australia; Letnick & Dickman, 2005). In others, such as the seasonal tropical savannas of northern Australia, the majority of species appear resilient to a range of fire regimes (Parr & Andersen, 2006). However, the questions of what is achievable across a particular landscape and what are the appropriate scale of patches and mix of regimes are difficult to answer, as highlighted by Wardell-Johnson *et al.* (2006). They described the intrinsic patchiness of fires that burned in particular landscapes, under particular climatic conditions, with a view to establishing operational guidelines for achieving a defined scale of mosaic.

What scale and pattern should be prescribed? Burrows & Abbott (2003) argued, as one of their "scientific principles to guide fire management" for conservation, that the *scale, or grain size, of the mosaic should (a) enable natal dispersal; (b) optimise boundary habitat (interface between two or more seral states); and (c) optimise connectivity (ability of fauna to cross between seral states)*. Many of the ecological processes I identified in 1995 as needing further study are relevant to the question of how the biota might respond to mosaics of fire ages or to mosaics of fire regimes (Table 2), including seed dispersal distances, patchiness and plant mortality, patchiness and plant-herbivore interactions, refugia and recolonisation of animals. Each species of organism may be unique in its ability to find, survive in and recolonise from refuges, and we cannot study each in turn. We may therefore have to predict responses to various mosaics from a limited set of life-history studies and then test these predictions with landscape-level experiments (see below).

ECOLOGICAL RESPONSES TO FIRE REGIMES

Inappropriate fire regimes have been recognised as potentially threatening to the conservation of biodiversity. Popular perceptions of what is "inappropriate" understandably focus on high-intensity fire, as in the comments by politicians quoted above. High-intensity fire certainly kills plants and animals and changes the 'look' of a landscape for years or decades, even centuries, in some ecological communities. In 1995, I argued that knowledge of the effects of high intensity fire on animal behaviour, mortality and source of re-establishment of populations was very scanty, and a recent review of fires in heathlands (Keith *et al.,* 2002) suggests that this is still the case, although the recent fires in 2001-02 and 2002-03 in south-eastern Australia are providing an opportunity for examining post-fire populations of plants and animals in sites of high fire intensity.

Frequency is another important element of fire regime in assessing inappropriate fire regimes. How frequent is too frequent? This is a difficult question to answer as a generalisation, because there is substantial variation from one region to another. In making predictions about the effects of fire regimes on the biota, Whelan *et al.* (2002) argued that the lack of empirical data made it necessary to infer responses from knowledge of life histories of the organism, other ecological processes, and characteristics of the fires, the landscape and the climate (Fig. 1). Using this approach, it is possible to use information on the time to first reproduction for obligate seeder shrubs to identify an inappropriate fire regime. The time to first reproduction for shrub species in south-western Australian Jarrah forests (Gill, 2002) appears to be as short as 2 years, but from about 1 to >9 years in Hawkesbury Sandstone woodlands (Keith, 1996). If these patterns are general within each region, a fire frequency of every four years might not cause local extinctions in jarrah forest, whereas fire intervals of less than 10 years would be expected to reduce biodiversity in Hawkesbury sandstone woodlands.

The box represents the life cycle of the organism, and the arrows represent attributes of the environment. (1) represents the processes determining survival, (2) represents the processes determining where colonists come from and when, (3) represents processes determining continued survival within the burned area, and (4) represents the processes determining rates of growth of individuals and the potential for

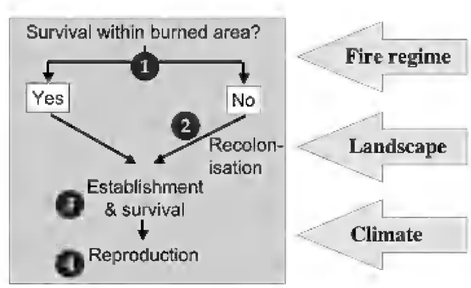

FIG. 1. Diagram of processes contributing to population change after fire (Whelan *et al.*, 2002).

reproduction and population increase.

A significant advance in the last decade has come in the area of defining the limits of tolerance of many plant species to extremes of fire regime. Because the empirical data are limited, these guidelines for ecological burning (e.g. Kenny *et al.,* 2003) are typically based on prediction from some of the key life-history characteristics, such as fire-sensitivity vs ability to sprout after fire, presence of a dormant vs transient seed bank, time to first reproduction. A similar approach should be possible with animals, and there have been some developments in this direction. For example, Friend & Wayne (2003) described the development of a framework for predicting fire responses of fauna based primarily on shelter, dietary and breeding requirements. Tasker *et al.* (2006) reviewed the published Australian literature on fire and fauna since 1995 and classified the studies according to the robustness of their design in terms of being able to infer cause-and-effect. This project will lead to the development of guidelines for ecological burning for fauna in NSW.

Approaches such as these are badly needed by land managers who have the dual responsibilities of protecting the neighbours outside the boundaries and protecting the biodiversity within. They are, however, sets of predictions – not empirical findings. As generalisations from those life-history characteristics that are considered to be "vital attributes" in the context of fire, they may not apply in all regions nor for all fires. It is critically important that the fragile ecological basis for guidelines such as these be acknowledged and that a process be developed for refining the knowledge for each particular location and learning whether the processes operating at one location for a functional group or individual species differ from those at another location. I consider that the next advance needed in fire ecology is the widespread development of an experimental approach to management, which is explored below.

EXPERIMENTS AND ADAPTIVE MANAGEMENT

In *The Ecology of Fire*, I included a chapter on approaches to fire studies, because I was strongly influenced by arguments of experimental ecologists, such as Tony Underwood (see Underwood, 1997). In the early 1980s, he asked me why fire ecologists concerned with the effects of different fire regimes on plant populations and communities had not manipulated fire regimes in replicated experiments. Many of the approaches used to infer fire effects are indeed flawed – and as scientists we should have known this for a long time: "No one would now dream of testing the response to a treatment by comparing two plots, one treated and the other untreated" (Fisher & Wishart, 1930 – cited in Underwood, 1986).

It is a sobering experience to review the papers on fire responses that have been published in the last 10 years and see how many infer a response to some aspect of fire based on a difference between two sites that experienced different fires. The important point here is not that such studies are worthless, because all ecological studies relating to fire contain important, hard-won observations. The issue is what inference is drawn from the observations. A finding of a statistically significant difference in mean seedling density in two sites, one burned in spring one year and the other burned the following autumn, can tell us only that the sites differ, no matter how much replication of quadrats we add, how well stratified we make them across each site, and how often we sample and for how long.

Parr & Chown (2003) presented an insightful summary of the components of a well designed fire ecology experiment – including appropriate scale, spatial replication, temporal replication, duration, and measurement of fire parameters. In reviewing research into fire and fauna in South Africa, they were unable to draw conclusions about the general effects of fire on the faunas of savanna, grassland or fynbos, because of the dearth of well-designed, well-replicated, comprehensive studies that test hypotheses about the ecological effects of fire. This is difficult for ecologists to accept, when so much effort is required even to gain this limited information. It is also difficult for managers, who are seeking certainty in conclusions about the effects of particular fire regimes in order to guide their fire management plans.

The Kapalga experiment in the Northern Territory was a landscape-scale fire experiment designed to test the effects of season of burning in tropical savannas on a range of elements of biodiversity (Andersen *et al.,* 2003; 2005). Experimental units were catchments 15-20 km^2, and fire treatments (early dry season, late dry season and unburnt) were replicated. The study was expensive to set up and maintain and ran for five years, which was sufficient in the tropical savanna habitat to have repeated fires in the treatment sites. A study

of this scale in temperate Australia, designed to test the effects of season and/or frequency of fires would need to continue for considerably longer and would probably be unsupportable in terms of continued resource demands.

There are good reasons for the dearth of well-designed, well-replicated, comprehensive studies at a large scale: they are expensive and difficult to conduct. There are trade-offs between the scale of the study and the amount of spatial replication. For example, a study completed several years ago (see Whelan & York, 1998 for the 1st instalment) was designed to test the effect of season of burning on post-fire recruitment of two bradysporous, obligate-seeder shrubs. We chose three replicate sites in which both species occurred, and in each site we set up four, 1-2 ha plots. We randomly assigned fires in each of two springs and two autumns to the four plots, and conducted (and contained!) the fires, with considerable input of resources by the Sydney Catchment Authority. Within each plot, we set up replicated locations into which we put 50 seeds, and applied two watering treatments – to test whether watering would offset any differences between seasons in recruitment. The reviewers of the manuscript argued that a major flaw in the study was the fact that the burned treatments were only 1-2 ha, and this scale issue was likely to be significant because of herbivory: herbivores were likely to concentrate in small burned plots thus elevating grazing pressure above what would be expected in a 'real' fire. This may be true, but larger experimental plots would have been out of the question unless we had been prepared to sacrifice some of the replication. Instead we included a grazing-exclusion treatment within the plots.

Good quality monitoring and comprehensive record-keeping in the past have allowed some researchers to design 'retrospective experiments', comparing aspects of biodiversity in replicated sites with different fire histories. Wittkuhn *et al.* (2008) shows how good CALM fire records in the Walpole region of WA, from 1972 to 2002 are being used to design studies that will test hypotheses about the impact of various between-fire intervals on biodiversity. Reasonable fire records over a >25 year time span enabled Cary & Morrison (1995) to use this approach to examine the impact of short between-fire intervals on the balance between obligate seeder and sprouter species in Sydney sandstone plant communities. Similarly, Burrows & Wardell-Johnson (2003) and Watson & Wardell-Johnson (2004) have used long-term fire records for sites with different fire histories (in the Jarrah forest region of WA and in south-east Queensland, respectively) to identify the plant species for which abundance was associated with frequently burned sites and those that were more abundant in sites burned less often. The Jarrah forest study was based on a long-term set of experimental burns in the "Lindesay Forest Block", in which season and frequency were manipulated. Measurement of fire responses (e.g. seedling density, survival) after a number of unrelated fires and at different times post-fire could be effective in testing the consistency of fire responses without needing complex, large-scale experimental treatments.

There appears to be quite a collection of long-term, manipulative fire experiments in Australia, many with relatively small plots, but nevertheless plots are replicated and fire regimes have been maintained. Given the resources needed to achieve this, it would be sensible to make more use of these experiments. What is needed is an accessible record of them across Australia, perhaps based on the information once collected by the Ecological Society of Australia to catalogue long-term ecological research sites (LTERs). The COAG Bushfire Inquiry (Ellis *et al.*, 2004) argued for the establishment of a national network of long-term ecological research sites to provide a basis for long-term monitoring of the impacts of fire regimes and fire events.

Although it may be unrealistic to expect landscape-level experiments to be set up in all major fire-prone ecosystems of Australia, land managers are conducting fires at a variety of scales, almost every year. How many of these are designed in collaboration with research staff, so that they can answer the very questions that land managers are asking of ecologists? An adaptive management approach to finding what fire regimes are appropriate for biodiversity conservation should have the following steps (Fig. 2): (i) make explicit the biodiversity objectives, (ii) recognise the lack of knowledge and clarify the questions that need to be answered, (iii) design burning prescriptions that can answer these questions, (iv) devise and fund monitoring and other data-collection activities, (v) review and communicate results, and (vi) use the new knowledge to modify the management prescription.

Adaptive management with these elements often meets with resistance from managers, because of the perceived delays, constraints imposed by needing to apply agreed treatments consistently, and costs

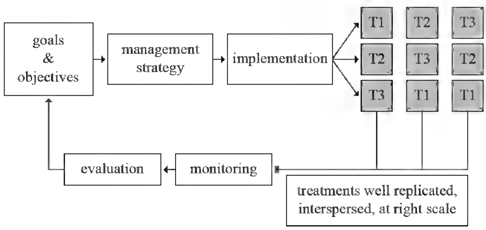

FIG. 2. Schematic diagram illustrating the steps involved in an adaptive management program (Whelan, 2003).

associated with monitoring. However, this seemed to me, in 1995, to be the only way in which fire managers will be able to know whether the burning prescriptions they are setting, based on ecological burning guides (themselves based on limited evidence), are actually maintaining biodiversity. There has been progress in the last decade, with a number of discussions of experimental approaches to management at conferences that include managers and scientists (e.g. the NSW Nature Conservation Council series – Gill, 2003), and a finding in the Report of the COAG Inquiry (Ellis *et al.,* 2004) supporting adaptive management as a way forward. The most recent example is the paper by Burrows *et al.* (2008), illustrating how such a program is being set up to determine the effects of fire management treatments on mainland Quokka populations. There is a good incentive for research ecologists to become involved in adaptive management in relation to fire – it might be the only way to get treatment plots at a sufficiently large-scale to make a reviewer happy!

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FUEL DYNAMICS AND FIRE SPREAD IN SPINIFEX GRASSLANDS OF THE WESTERN DESERT

N.D. BURROWS, B. WARD AND A. ROBINSON

Burrows N.D., Ward B. & Robinson A. 2008 06 25: Fuel Dynamics And Fire Spread In Spinifex Grasslands Of The Western Desert. *Proceedings of the Royal Society of Queensland*, 115: 69-76. Brisbane. ISSN 0080-469X.

Spinifex grasslands, characterised by the dominance of perennial hummock grasses of the genus Triodia, cover some 43% of Western Australia. The combination of the physical structure of the vegetation and the often extreme weather conditions makes spinifex grasslands highly flammable. Historically, lightning and burning by Aboriginal people maintained much of the grasslands as a fine-grained mosaic of vegetation at different seral stages providing diverse habitats. Where traditional Aboriginal burning has ceased, large summer wildfires have reduced habitat diversity. In many areas there is a need to reintroduce anthropogenic burning to re-establish habitat diversity at finer scales as well as to protect human life, property and cultural values. An understanding of fuel dynamics and fire behaviour is fundamental to managing fire in these landscapes. Spinifex grasslands have the potential to re-burn 5-7 years after fire, depending on rainfall, but can take 18-20 years to reach full maturity as a fuel. Because of the discontinuous nature of the vegetation, there are thresholds of combinations of fuel load, fuel moisture and wind speed that need to be exceeded before fire will spread. Once thresholds have been exceeded, fires have the potential to spread rapidly, especially under high wind speeds. In this paper we present a revised empirical model for predicting threshold fuel and weather conditions for fire initiation and subsequent rate of spread over a broad range of fuel and weather conditions.

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There is now compelling evidence that the fire regime in much of the spinifex grasslands of the Western Desert region of Australia has changed with the cessation of traditional Aboriginal burning from a fine grain mosaic of burnt patches at different seral states to a coarser grain, simplified mosaic of infrequent, large wildfires (Burrows & Christensen, 1991; Burrows *et al.,* 2006). This, together with predation by introduced predators, has probably contributed to the alarming decline in native fauna, particularly medium sized mammals and some ground-nesting birds (Johnson *et al.,* 1989; Burbidge & McKenzie, 1989; Morton, 1990). Management intervention by the reintroduction of fire to re-create a fine-grained mosaic of different post-fire seral states and to fragment wildfires, is a desirable strategy in many areas. However, fire management is constrained by limited resources, the vastness and remoteness of many nature conservation reserves, poor accessibility and imperfect knowledge of fire behaviour and fire effects. An understanding of fuel dynamics, especially changes in the structure and quantity of spinifex grassland fuels with time since last fire is important for understanding fire hazard, fire behaviour and for planning prescribed burning operations. This study aimed to improve an understanding of fuel dynamics and to improve the predictability of fire behaviour, particularly threshold conditions for fire start and spread, over a wide range of fuel and weather conditions.

MATERIALS AND METHODS

The spinifex grasslands fire spread model reported here was derived from field experiments undertaken in the two remote regions of Western Australia described below. Modelling incorporated data from 41 experimental fires previously conducted in the Gibson Desert and reported by Burrows *et al.* (1991). These data were supplemented by a further 42 experimental fires carried out in the Great Sandy Desert over the period 1992-1994.

FIELD SITES

Fuel dynamics and fire behaviour studies were conducted in hummock grassland communities in the Gibson Desert (Gibson Desert Nature Reserve), Little Sandy Desert (Lorna Glen ex-pastoral lease) and Great Sandy Desert (Rudall River National Park) in Western Australia. A description of the Gibson Desert study area

(~location 24° 44' S latitude and 124° 44' E longitude) is provided by Burrows *et al.* (1991). In summary, and as described by Beard (1969), the Gibson Desert is 'characterised by laterite plains, a monotonous and gently undulating topography floored with ironstone gravel and vegetated with poor spinifex (mostly *Triodia basedowii* and *T. schinzii*) and stunted mulga (*Acacia anuera*)'. Spinifex communities growing on the stony plains in the Gibson Desert are generally sparser, shorter and lower in biomass ('poorer') than those that occur on the sand plains and dune fields that characterise much of the inland arid zone, including the Great Sandy Desert. The Rudall River National Park is situated in the south-western portion of the Great Sandy Desert with the centre of the Park being at approximately 22° 30' S latitude and 125° E longitude. The climate of the region is classified as desert based on the Koppen classification system. Mean annual rainfall, while highly variable, is ~225 mm and mainly derived from summer thunderstorms and cyclone activity between November and March. Mean daily maximum temperatures range from about 25° C over the winter months to about 41° C over the summer months. Vegetation on the red eolian sand plains and dune fields of the study area is dominated by spinifex (*Triodia pungens, T. wiseana* and *T. schinzii*), with scattered small shrubs and trees (predominantly species of the genera, *Grevillea, Acacia, Hakea, Eremophila* and *Eucalyptus*). Spinifex cover mostly varies from 30-50%, ranging in height from 0.2-0.4 m, with scattered trees and shrubs to 3 m. The Lorna Glen study site in the Little Sandy Desert was located in hummock grassland (predominantly *Triodia basedowii*) on red eolian sand plains with scattered low woody shrubs and tree and mallee *Eucalyptus* species. The climate of the region is classified as desert with a mean annual rainfall of ~220 mm. Mean daily maximum temperatures range from about 20° C over the winter months to 38° C over the summer months.

The experimental methods employed to develop the model are similar to those published by Burrows *et al.* (1991), so will be summarised here. Prior to igniting the fires, measures of the structure (height, cover and patchiness) and biomass of the spinifex grassland to be burnt were made by intensive sampling along a series (3-4) of 100 m continuous line transects. Spinifex grasslands are usually a simple (single layer) elevated fuel dominated by hummocks of spinifex of more-or-less uniform structure (for a given time since last fire) interspersed with low soft grasses and occasional small shrubs and trees. The distance along the line of each of bare ground, spinifex hummock, soft grasses and other vegetation intercepted by the line, was recorded in continuous sequence along each transect. Where vegetation was intersected along the transect, it was categorised as either spinifex, soft grasses, herb, litter, woody shrub or 'other', and the contiguous distance of the cover type along the transect, and its height above ground, was recorded. This enabled vegetation cover (% by vegetation classes) and proportion (%) of bare ground, and patch dimensions (fuel patchiness), to be quantified. These measurements of vegetation cover and patchiness, together with fuel load, fuel moisture content and wind speed, were used to model fire behaviour. Fuel load (oven dry weight) was sampled by removing all fine (<6 mm diameter) live and dead vegetation from 1 m × 1 m quadrats placed at 10 m intervals along the 100 m line transects. To gain an understanding of post-fire fuel (biomass) increment, additional fuel studies using the method described above were carried out in spinifex grasslands at Lorna Glen, an ex-pastoral lease in the southern part of the Little Sandy Desert some 180 km NE of the town of Wiluna, Western Australia. A range of different but known fuel ages (time since last fire), varying from 2 years to 42 years, were sampled by assessing up to 4 × 100 m line transects in each fuel age.

For the fire behaviour studies, moisture content of the spinifex hummocks was determined by taking a cross section (profile) of material from 6-10 hummocks prior to each experimental fire and determining moisture content by oven drying the samples at 80° C for 48 hours. Weather conditions during the experimental fires were recorded by both an on-site automatic weather station and hand held instruments. Wind speed, wind direction, air temperature and relative humidity were measured at ~2 m above ground (see Burrows *et al.,* 1991). Experimental fires were lit by a 100-200 m line of fire set at right angles to the wind direction. The position of the fastest spreading part of the fire (headfire) was marked at regular time intervals with metal tags, which were later surveyed to enable the fire's rate of spread to be calculated. Not all fires sustained spread and analysis focussed on; a) determining threshold conditions of fuel and weather for fire spread and b) predicting rate of spread when these were exceeded.

ANALYSIS

Models were developed by applying various linear and non-linear regression techniques to determine the best statistical relationships between dependent

(fire behaviour) variables and independent (fuel and weather) variables. Logistic regression modelling (SAS, 2002), a conditional distribution of a binary output variable given a range of input vectors, was used to determine the probability of fires spreading.

RESULTS AND DISCUSSION

Space-for-time fuel sampling showed that fuel load increased with time since fire and plateaued at about 10-12 t ha^{-1} some 18-20 years after fire (Fig. 1). Changes in the cover of spinifex, other plant species and of bare ground with time since fire are shown in Figure 2. In the early years post-fire, vegetation cover was very sparse, although on some sites, and following rain, the cover of annual herbs and grasses temporarily increased the overall cover of vegetation. Figure 2 shows the increasing dominance of spinifex cover with time since fire, which reached a maximum of about 40% some 18-20 years after fire. By this stage, other plant species comprised about 5% of the total cover, with some 55% of the sand plain being bare ground. The mean height of spinifex clumps also steadily increased with time, plateauing at about 35 cm some 18-20 years after fire (Fig. 3). This pattern of post-fire fuel development is consistent with other studies (Griffin, 1991; Griffin *et al.,* 1990; Allan & Southgate, 2002), which also note that the rate of change in biomass and fuel composition and structure are strongly influenced by rainfall. The profile moisture content of spinifex clumps of varying age, but each sampled at the same time, decreased with the age of the clump, reflecting the increasing proportion of drier dead material and decreasing proportion of moister, green live material in the hummocks (Fig. 4). These data demonstrate how the flammability of spinifex grasslands increases with time, not only due to increasing cover and load of flammable vegetation, but also due to decreasing moisture content and

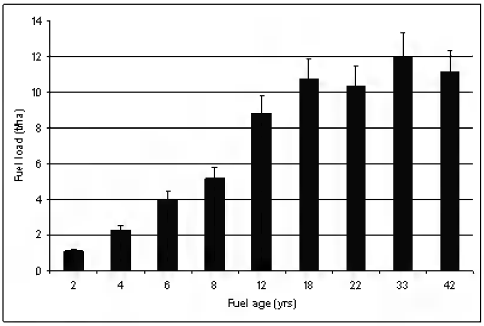

FIG. 1: Fuel load (oven dry weight) with fuel age (time since last fire) in spinifex grassland.

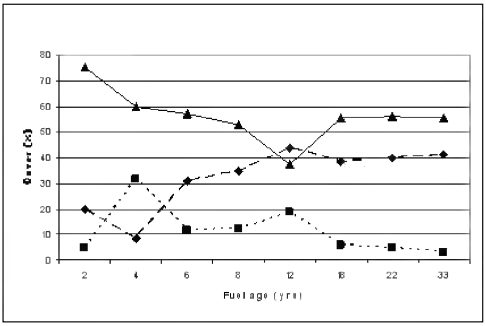

FIG. 2: Cover of vegetation and of bare ground with fuel age (time since last fire) in spinifex grassland. Triangle = ‘bare ground’, square = ‘other’, diamond = ‘spinifex’.

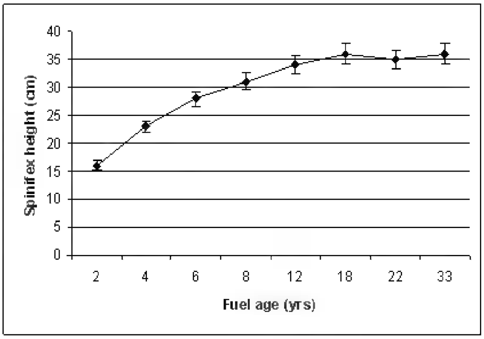

FIG. 3: Change in the mean height of spinifex hummocks with fuel age (time since last fire).

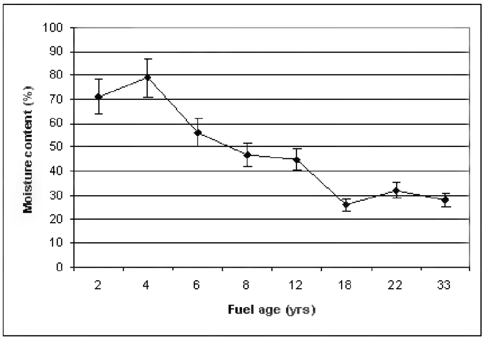

FIG. 4: Spinifex hummock profile moisture content, measured at a fixed point in time, with hummock age (time since last fire).

an increasing proportion of drier dead material accumulating in the hummocks. Of the 100-200 m of ignition line used to start the fires, rarely did the fire progress in a continuous line of headfire. Commonly, 'tongues' or 'fingers' of headfire developed from the ignition line, with most being <100 m wide.

The range of fuel, weather and fire behaviour conditions experienced during the fire behaviour experiments is shown in Table 1. Gill *et al.* (1995) suggest three stages in the formulation and application of fire spread models; (i) a domain analysis for the applicability of inputs to a fire spread model, (ii) a likelihood of fire spread analysis and (iii) application of a spread model to predict rate of spread. In discontinuous or patchy fuels such as spinifex grasslands, there are multiple thresholds to fire spread (Burrows *et al.,* 1991; Gill *et al.,* 1995). From the outset, conditions of fuel and weather needed to be such that the flame dimensions are sufficient to breach gaps in the fuel. Therefore, an important step in predicting fire behaviour is to determine the probability that, following ignition, fire will actually spread for a given set of conditions. Fuel moisture content is the main factor limiting ignition and sustained fire spread in continuous fuels, but in discontinuous or patchy fuels, such as hummock grasslands, fire spread can only be sustained if conditions are such that the flames from burning hummocks can breach the inter-hummock gaps and ignite the adjacent hummock (Gill *et al.,* 1995). Factors that determine fire energy, flame size and flame tilt, therefore the capacity for sustained spread, include wind speed (and slope), fuel load, fuel moisture content and fuel structural characteristics such as height, cover and patchiness (or separation of fuel pieces).

TABLE 1: Experimental conditions under which the spinifex grassland fire spread model was developed (n = 83 fires).

Variable	Mean	Range
Wind speed (km h^{-1})	15	4 – 36
Temperature (°C)	31	19 – 50
RH (%)	14	5 – 48
Fuel load (t ha^{-1})	7	2 – 14
Fuel cover (%)	38	9 – 65
Fuel height (cm)	25	18 – 37
Fuel profile moisture (%)	18	12 – 31
Rate of spread (m h^{-1})	842	0 – 5,520
Flame height (m)	1.4	0 – 5
Fire intensity (kW m^{-1})	3,515	0 – 19,111

Of the independent variables measured, wind speed, fuel load and fuel moisture content were found to be the most important variables influencing whether or not fires would spread. Fuel load is important in its own right, but is also a surrogate for, and correlated with cover and height. Using these factors, the probability of fire spreading was best estimated by an applied logistic function (SAS, 2002) of the form:

$$SI_{FL} = 0.57(W) + 0.96(FL) – 0.42(PMC) – 7.42$$
(Equation 1), or,

$$SI_{FF} = 0.37(W) + 0.78(FF) – 0.31(PMC) – 5.23$$
(Equation 2).

Where:

SI_{FL} = Fire Spread Index incorporating fuel load. A positive value means fire will probably spread.

SI_{FF} = Fire Spread Index using fuel factor (see below).

W = average wind speed (km h^{-1}) over a 5 minute period at 2 m above ground.

FL = fuel load (spinifex and other fine fuels) (oven dry weight in t ha^{-1}).

PMC = the profile moisture content of the spinifex hummock (% oven dry weight).

FF = fuel factor = 0.25(CV) + 0.04(HT) - 3.2
(Equation 3) (R^2=0.71)

Where:

FF = fuel factor (a surrogate for fuel load incorporating fuel cover and height)

CV = fuel (spinifex) cover (%)

HT = mean hummock height (cm)

Fuel load (FL) and fuel factor (FF) are related by the equation:

FL = 0.98(FF) - 0.08 (Equation 4) (R^2=0.71).

FF is almost equal to FL so can be substituted for FL if fuel load cannot be measured in the field. Equation 3 assumes a more-or-less constant hummock bulk density of about 17 kg m^3. This can vary within and between species depending on sight (e.g., soil type and termite activity) and seasons (rainfall), so fuel load is the preferred variable, hence Equation 1 the preferred equation for predicting the likelihood of fire spread.

INTERPRETING THE FIRE SPREAD INDEX (SI)

SI (Equations 1 & 2) is an applied logistic regression function, which means that the outcome is binary, or dichotomous. That is, it determines whether or not fire will spread. If the SI is negative, then fire should not

TABLE 2: The likelihood of fire spread (the Spread Index - SI) in spinifex grasslands, and potential rates of spread when the threshold conditions for spread are exceeded (SI>0).

SI	Likelihood of fire spread	Fire Danger	Potential ROS (m h^{-1})
SI ≤ -2	Fire unlikely to spread	Very Low	0
-2 < SI ≤ 0	Fire may spread	Low	<500
0 < SI ≤ 2	Fire should spread	Moderate	500-900
2 < SI ≤ 4	Fire will spread	High	900-1,800
4 < SI ≤6	Fire will spread	Very High	1,800-2,700
6 < SI ≤10	Fire will spread	Extreme	2,700-4,500
SI > 10	Fire will spread	Very Extreme	>4,500

spread; if it is positive, then fire should spread. The more negative the value, the less likely is spread and vice versa (see Table 2).

PREDICTING RATE OF SPREAD

Having determined whether or not a fire will spread (SI), the next step involved developing a model to predict rate of spread. Once conditions are suitable for fire spread, of the variables measured, rate of spread was found to be best explained by a linear function of wind speed, fuel load and fuel moisture content, with wind speed being the most influential variable (Equations 4 & 5 below and Fig. 5). Other variables such as fuel cover and height were also found to be important, but these are related to, and are best represented, by fuel load. In essence, rate of spread in discrete fuel arrays depends on flame dimensions which in turn depends on wind speed, fuel load and fuel moisture content. Clearly, a feed back mechanism exists, as more fuel becomes involved at higher rates of spread, which in turn results in larger flames. Two (linear) prediction models were developed; one requires measurement of fuel load (dry weight), the other requires the more easily obtained measures of fuel cover and height. Temperature and relative humidity were found not to be highly significant, so were not included in the model. It is important to note that these fire spread models do not apply when conditions are below thresholds for fire spread, i.e., SI < 0. The relationship between actual rate of spread (of experimental fires) and that predicted using Equation 4 is shown in Figure 6.

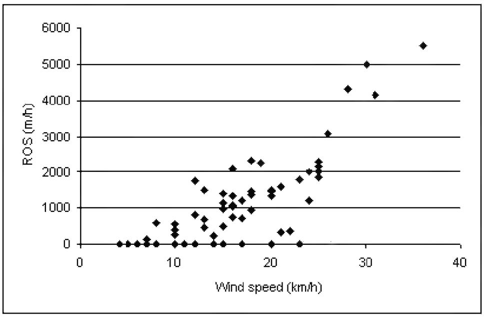

FIG. 5: Headfire rate of spread in spinifex grassland with wind speed measured 2 m above ground.

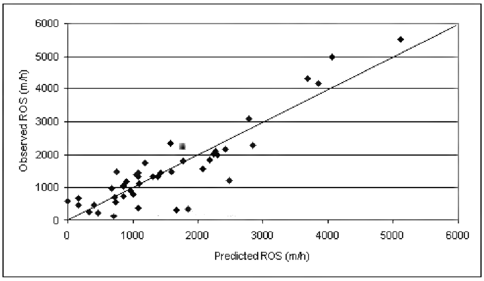

FIG. 6: Relationship between observed and predicted rate of spread of experimental fires in spinifex grassland.

Forward Rate of Spread (ROS):

$$ROS_{FL} = 154.9(W) + 140.6(FL) - 228.0(PMC) + 1581$$
Equation 4 (R^2=0.79)

Or;

$$ROS_{FF} = 142.8(W) + 120.1(FF) - 229.1(PMC) + 1969$$
Equation 5 (R^2=0.79)

Where;

ROS_{FL} = forward rate of spread calculated using fuel load (m h^{-1})

ROS_{FF} = forward rate of spread calculated using fuel factor (m h^{-1})

W = average wind speed (km h^{-1}) over 5 minutes @ 2 m above ground.

FL = fuel load (spinifex and other fine fuel) (oven dry weight in ha^{-1}).

PMC = the profile moisture content of the spinifex hummock (% oven dry weight).

FF = fuel factor (see above).

MEASURING MODEL INPUT VARIABLES

The models described above will perform best if the input (independent) variables are measured the same way as they were measured for the experimental fires on which the models are based. Alternative models for predicting fire behaviour (thresholds for spread and rate of spread) are presented above so that fire managers can choose the most practical model for their circumstances. For example, in some cases it may be easier to measure fuel cover and height (surrogates) than to measure fuel load. Using surrogate variables will reduce the reliability of the models.

MODEL LIMITATIONS AND UNCERTAINTIES

Combustion and bush fire behaviour are complex phenomena that are poorly understood at the fundamental level. The empirically-derived spread models presented here do not include all of the potential variables likely to influence fire behaviour, but include key integrator variables that are relatively straight forward to measure in the field. The models explain between 70% and 80% of the variation in the observed spread rates of experimental fires burning in discontinuous spinifex grassland fuels. The models are constrained by the parameter bounds described in Table 1, which represent a wide range, but not all, of potential burning conditions likely to be encountered in hummock grasslands. The experimental fires on which the models are based did not capture the entire range of fuel structure, fuel moisture, weather and terrain conditions likely to be experienced or found in hummock grasslands throughout Western Australia. For example, an obvious range/variable omission is the influence of slope and slope-wind interactions, which are likely to be important in some regions. Where slope is an important terrain variable, then we recommend adopting the formula developed by Burrows (1994) for forest fuels as a guide. Roughly, rate of spread (upslope) doubles for every 10º of slope (McArthur, 1967). That is;

$$ROS_{SC} = ROS^{*}e^{(0.0687S)}$$

Where;

ROS_{SC} = rate of spread corrected for slope (m h^{-1})

ROS = rate of spread on flat terrain (m h^{-1})

S = slope (degrees)

The models were developed using line ignitions, which may not be the preferred ignition technique in prescribed burns. It is more likely that aerial incendiaries (point ignition) will be used in prescribed burn operations in remote or poorly accessible areas. Cheney *et al.* (1993) working in more-or-less continuous annual grassland fuels demonstrated that initial rate of spread was significantly influenced by the length of the ignition line. This was not tested here but in discontinuous fuels such as spinifex vegetation, the length of effective ignition line and subsequent width of the headfire is likely to influence both the probability of sustained ignition and rate of spread in the early stages post-ignition.

Spinifex hummocks consist mostly of live vegetation and have the capacity to persist at very low moisture contents. The proportions of live and dead material in the hummocks varies and depends on species, age (time since last fire), seasonal conditions and termite activity. Average variation in the live-to-dead fuel ratio is not accounted for by the models. These models do not take account of fire propagation by spotting. Except in the presence of mallees, other eucalypts or other myrtaceous scrub when spot fires were observed to start up to 800 m from the main fire, we observed only short distance spotting (< 100 m) and spot fires were usually quickly overrun by the main headfire. The fuel continuity of hummock grasslands can change significantly following heavy rainfall events. The normally bare ground between the hummocks can become overgrown or partially overgrown with soft grasses and other annuals, forming a more-or-less continuous fuel. When this occurs, the models presented here will underestimate fire spread thresholds and rates of spread.

CONCLUSIONS

The fire regime has changed significantly and recently in parts of the Western Desert following the departure of Aboriginal people and the cessation of traditional burning. Today, and in the absence of regular burning by Aborigines, fuels have accumulated over vast areas and when lightning (or people) ignites these fuels and under hot, dry, windy summer conditions, large and intense wildfires occur. It is imperative that anthropogenic fire is reintroduced into these landscapes to prevent further declines in ecosystem health. The spinifex grassland fire behaviour models presented here can be used to assist fire and land managers plan and implement prescribed fire and make better assessments of fire danger and of wildfire behaviour to aid pre-suppression and suppression actions.

ACKNOWLEDGEMENTS

We acknowledge and thank Canning Resources Pty Ltd. based at Kintyre and Bruce Harvey (CRA exploration geologist) for invaluable support for this work. We also thank Pilbara and Goldfields Regional staff of the Department of Conservation and Land Management (CALM) who provided logistic support and for keeping vigil on HF radio. This work is dedicated to the memory of Alex Robinson, now deceased.

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PREDICTING THREATENED SPECIES RESPONSES TO FUEL REDUCTION FOR ASSET PROTECTION

R.J. WHELAN, L. COLLINS & R. LOEMKER

Whelan, R.J., Collins, L. & Loemker, R. 2008 06 25: Predicting Threatened Species Responses To Fuel Reduction For Asset Protection. *Proceedings of the Royal Society of Queensland*, 115: 77-83. Brisbane. ISSN 0080-469X.

Fuel reduction in bushland adjacent to urban development is an important component of bushfire management to protect lives and properties. In many urban areas, the objective of property protection by fuel reduction conflicts with biodiversity management objectives. Conserving threatened species in such situations will require information on spatial distributions of these species in the landscape. We used GIS modelling to predict the likely impacts of strategic fire advantage zones (SFAZs) on two threatened species in the Shoalhaven region of NSW: the eastern bristlebird and the glossy black cockatoo. We used current knowledge of the association between these animals and vegetation to predict habitat suitability, overlaid residential areas on this habitat-suitability map, and then applied buffers around the residential areas to represent minimum (250 m) and maximum (450 m) SFAZs scenarios. For eastern bristlebirds, 4,000 ha of suitable habitat occurred in the study area, and nearly 5% of this would become unsuitable with 450 m SFAZs. For the cockatoos, approximately 9% of 30,000 ha of suitable habitat would be altered with 450 m SFAZs. The GIS models provide the information needed for more creative bushfire mitigation activities that could deliver both the conservation of endangered species and protection of human assets.

❒ fire, eastern bristlebird, glossy black cockatoo, GIS, habitat model

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Conflicts occur between biodiversity conservation and the need to protect life and property against bushfire at the urban/bushland interface, making it the focal point of much debate (Nature Conservation Council, 1999; 2002; 2004). A risk-management approach is an increasingly common strategy for dealing with this conflict (Ellis *et al.*, 2004) and defining bushfire risk zones in bushland surrounding urban areas is a key part of this approach. In NSW, risk to property is addressed largely through Asset Protection Zones (APZs) and Strategic Fire Advantage Zones (SFAZs), which are managed so that fuel loads remain below defined levels (RFS, 2003).

Specifications for APZs in the Shoalhaven region allow a maximum width of 100 m, with mechanical clearing usually between 20-40 m to maintain fuel loads below 4 t ha^{-1}. Specifications for SFAZs vary. Generally, the combined width of SFAZ and APZ would be 400 m, but may extend further, depending on local conditions. Fuel loads in SFAZ are to be kept to a maximum of 7-10 t ha^{-1} over 60-80% of the area, usually by prescribed burning (RFS, 2003).

Clearing of native vegetation can have dramatic impacts upon native flora and fauna, causing loss of species locally, fragmentation of habitats, and disruption of ecosystem processes (Saunders, 1977, 1990; NPWS, 2006). Similarly, deliberate burning at high frequency can have negative effects upon native biota (Brooker, 1998; Baker, 2002; Brown *et al.*, 2003; NPWS, 2004). The detrimental ecological impacts of clearing and altered fire regimes have led to these being listed as Key Threatening Processes under NSW threatened species legislation (TSCA, 1995).

A number of species listed as vulnerable and endangered under NSW legislation are identified as being at risk of extinction when exposed to high-frequency fire, as it may disrupt their life cycle processes and make the vegetation structure of their habitat unsuitable (NPWS, 2004). Four characteristics in particular appear to make animal species susceptible to high-frequency fire and resultant habitat change: ground dwelling, cover-dependant, poor dispersal and low fecundity (Keith *et al.*, 2002; Whelan *et al.*, 2002).

The Shoalhaven region encompasses 4,660 km^2, and is a very important area ecologically due to its wide range of habitat types and high biological diversity. The Shoalhaven Local Government Area (LGA) is under significant pressure for further development but some parts of the landscape are dedicated to conservation in national parks. In addition to the city of Nowra, there are many small urban and rural settlements, which create long urban/bushland perimeters. Within the Shoalhaven region, there are 36 species of plants and 89 species of animals listed in Schedules 1 and 2 of the New South Wales Threatened Species Conservation Act, 1995 (SCC, 2004).

We identified seven of the listed animal species that are likely to be sensitive to high frequency fire: Spotted Tailed Quoll (*Dasyurus maculatus*), Ground Parrot (*Pezoporus wallicus*), Long-nosed Potoroo (*Potorous tridactylus*), Southern Brown Bandicoot (*Isoodon obesulus*), Squirrel Glider (*Petaurus norfolcensis*), Eastern Bristlebird (*Dasyornis brachypterus*) and Glossy Black Cockatoo (*Calyptorhynchus lathami*). Our aim was to develop models of habitat suitability and quantify the spatial overlap between suitable habitat and SFAZs. There were sufficient location records in the Shoalhaven region to allow habitat suitability assessment for only two of these species, the Eastern Bristlebird and the Glossy Black Cockatoo. We focused upon the north eastern part of the Shoalhaven LGA, between Nowra in the north and Manyana in the south, because this area contained both a significant number of records of Eastern Bristlebirds and Glossy Black Cockatoos and also a large number of urban/bushland interfaces.

MATERIALS AND METHODS

STUDY SPECIES

The Eastern Bristlebird is a small, semi-flightless terrestrial bird, with a sparse distribution that is confined to south east Queensland and lower parts of south east Australia (Baker, 1997). Two of the largest and most significant populations occur in south east NSW at Jervis Bay and Barren Grounds-Budderoo (Baker, 2002). The Eastern Bristlebird occupies a broad array of vegetation communities, each characterised by dense ground cover, which appears to be an important habitat requirement (Baker, 2000; 2002). This species displays all of the attributes of a fire sensitive bird, which hinders its ability to escape fire (ground dwelling, poor flier) and recover afterwards (cover dependant, poor disperser, low fecundity). The suitability of habitat post fire for this species will be dependant upon the time it takes for vegetation to recover and attain an appropriate structure.

The Glossy Black Cockatoo is the smallest of the Black Cockatoos, and is patchily distributed along the coast between Eungella in Queensland to Mallacoota in Victoria, with scattered populations being located in central south Queensland and NSW, and an isolated population located on Kangaroo Island South Australia (Higgins, 1999). The main habitat requirement for this species is the presence of *Allocasuarina* species for food, with the preferred *Allocasuarina* species in coastal south east NSW being *A. littoralis* (Clout, 1989; Mills, 1996). The presence of hollow bearing *Eucalyptus* trees for nesting is also very important (Joseph, 1982). The response of the Glossy Black Cockatoo to fire has not been the subject of detailed study, but Collins (2005) found that the number of trees showing evidence of foraging, and the number of cones consumed per tree increases with time since fire, peaking between 15 and 20 years post fire. The number of trees per hectare with evidence of feeding has been shown to be highly correlated with the number of Glossy Black Cockatoos in an area (Pepper, 1997).

GIS MODELLING

The use of GIS as a management tool is becoming increasingly common in conservation biology and land management planning, as it allows real world phenomena to be approximated using spatial relationships between geographical data in the computer-based environment (Delaney, 1999), and also allows effects of various scenarios to be tested at a landscape scale. For each species, the area of suitable habitat that would be affected the fuel-reduction burning was calculated in two simulations, one assuming a 250 m wide SFAZ and the other 450 m wide.

In each case, the appropriate SFAZ layer was intersected with a 'suitable habitat' layer constructed for each species. Suitable habitat layers were created by (i) identifying key habitat requirements of each species from literature and (ii) selecting appropriate vegetation units from the vegetation map layer obtained from the NSW Department of Environment & Climate Change (DECC) based on these suitable habitat types and key habitat requirements using descriptions of each vegetation unit, which contained information about groundcover and shrub density and dominant plant species (Tindall *et al.*, 2004; Table 1).

We used records from the Atlas of NSW Wildlife (5 Sept. 2005), NSW DECC, to estimate the accuracy for the 'suitable habitat' models for each species. Each record within 500 m of a road was classified as being in predicted suitable habitat or not. We used the 500 m buffer because most survey effort has clearly been concentrated near roads. Although there were some records at greater distances, we considered that survey effort has been very sporadic away from roads. For Eastern Bristlebirds, 70% of the 263 records within 500 m of roads were in sites classed as suitable habitat; for Glossy Black Cockatoos, 65% of 253 records were in suitable habitat.

Minimum and maximum SFAZ layers were created by placing a 250 m and 450 m buffer around the villages in the study area. These buffers were edited so that they did not extend across significant water bodies. Habitat models were then intersected with an SFAZ data layer to identify the area of habitat affected by SFAZs for both the minimum and maximum widths. This was then exported as a new layer of suitable habitat impacted upon by SFAZs. The 'calculate perimeter and area' tool in X Tools Pro 2.0 (Data East, LLC) was then used to calculate the total area of 'suitable habitat' in the study region, and the portion of suitable habitat that was within SFAZs for each simulation.

RESULTS AND DISCUSSION

HABITAT SUITABILITY

The habitat model predicted that 90,000 ha is habitat suitable for the Eastern Bristlebird, and 30,000 ha is suitable for the Glossy Black Cockatoo within the 166,000 ha study region. This study region is particularly important for these threatened bird species. In particular, it represents one of the only two substantial, viable populations of the endangered Eastern Bristlebird remaining – from a species distribution that once spanned coastal areas from northern Victoria to southern Queensland (Baker, 1997; 2000).

The habitat model was not perfect; a number of records for each species were in areas classified as 'unsuitable' habitat in the model. There are several possible reasons for such misclassification, some associated with habitat modelling and some with bird records. For habitat modelling, these include: (i) the information for the vegetation layer in the GIS is based on aerial photograph interpretation, limited field sampling and vegetation distribution modelling from environmental data layers, so some areas may have been identified as unsuitable when, in the field, they may have been suitable habitat; (ii) the scale and resolution of the mapping may have led to mis-identification of small (<1 ha) remnant patches of suitable habitat as unsuitable, even though birds may have been able to use such small patches; (iii) estimation of % cover of low and mid-storey vegetation, which is critical for Eastern Bristlebirds, is subjective and this vegetation characteristic may change rapidly, especially as a result of fire. For bird records, sources of misclassification include: (i) the records for the two bird species in the DECC Wildlife Atlas represent an accumulation spanning many decades; (ii) older records that now appear to be misclassified may have actually been in what was then suitable habitat; (iii) some records may have been due to birds being detected while they were moving through habitat, but not primarily using it; (iv) in the case of the Eastern Bristlebird, many may be in close proximity to suitable habitat, reflecting the fact that this species occurs at higher density in areas of vegetation transition (Baker *et al.*, 2002) – the fact that records in unsuitable habitat are generally close to suitable habitat can be seen in Figure 2a.

Despite these various possible sources of misclassification, 70% of the Eastern Bristlebird records and 65% of Glossy Black Cockatoo records occurred within areas classed as suitable habitat. Thus, we consider that even this relatively coarse habitat modelling can give a good representation of the areas of potential conflicts between

TABLE 1. Vegetation units (from Tindall *et al.,* 2004) in Shoalhaven GIS layer that were classed as suitable habitat in habitat modelling.

Attribute	Eastern Bristlebird	Glossy Black Cockatoo
Suitable vegetation types	Wet forest/rainforest, forest, open forest, woodland, open woodland, closed scrub, closed coastal scrub, closed shrub swamp, mallee, closed low to tall heathland, closed sedgeland and closed wet heathland (Higgins, 1999; Baker, 2000)	Woodland or open sclerophyll forests (Higgins, 1999)
Key habitat requirements	Dense groundcover (Baker, 2000; 2002)	Presence of *A. littoralis* trees (Clout, 1989; Mills, 1996)
Requirements for habitat modelling	Any of the above vegetation units with >34% groundcover	Woodland or open sclerophyll forests with *A. littoralis*

fire management and biodiversity management for these two threatened species.

SFAZ IMPACTS

The distribution of suitable habitat across the landscape in the study region results in varying amounts falling within SFAZs, depending both on the species and on the width of SFAZ in a particular scenario. The potential magnitude of the impact of habitat conversion from suitable to unsuitable as a result of maintaining SFAZs ranges from 1,343 to 2,500 ha for Glossy Black Cockatoos (4.5 to 8.3% of their total suitable habitat) and from 2,150 to 4,000 ha for Eastern Bristlebirds (2.4 to 4.5% of their suitable habitat) (Table 2).

Figure 1 illustrates both the distribution of suitable habitat modelled for these two species, and the overlap with the 450 m SFAZ buffer. This figure clearly illustrates the long urban/bushland perimeters that are associated with having a number of small urban settlements within a bushland landscape, and the large areas that need to be managed for fuel-reduction as a consequence.

EFFECT OF LANDSCAPE CONTEXT

The modelling conducted in this project represents a snapshot in time. The landscape is not constant, however. First, large-scale phenomena such as climate change will alter the composition of vegetation communities and their distribution across the landscape, hence changing the amount and spatial distribution of suitable habitat. Second, further urban and other development will continue to reduce the total amount of suitable habitat in the region both directly, by conversion from bushland to urban, and indirectly, by further increasing the urban/bushland perimeter and hence to total area of the landscape in SFAZ. Third, periodic unplanned fires will temporarily convert suitable habitat to unsuitable; in fact, bushfires between 2000 and 2003 burned 88,500 ha in the study region (Loemker, 2004), converting 60.6% of formerly suitable habitat to unsuitable for Eastern Bristlebirds. As a result, the percentage of remaining Bristlebird habitat that was contained within SFAZs increased from 4.5% to over 10%.

An important consequence of conversion of habitat from suitable to unsuitable, as a result of further development, bushfires, or SFAZ management, is increased fragmentation and separation of remnant patches of suitable habitat. For example, maintenance of the SFAZ around Wright's Beach (the small settlement in the middle-lower section in Figure 2a), would separate suitable Bristlebird habitat to the south from a large tract to the north. The spatial arrangement of this remnant 'patchwork' will influence both viability of local populations in the remnants and their ability to recolonise patches that become suitable as post-fire regeneration occurs. Land managers, as well as urban and regional town planners, need to consider these factors when managing urban development.

The value of small patches of suitable habitat and the degree of impediment to recolonisation that is caused by converting habitat to unsuitable is dependent upon species. Glossy Black Cockatoos, for example, are more mobile and wide ranging than Bristlebirds and are less dependent on cover, though they are perhaps more dependent on a particular food source (Allocasuarina seeds). Their mobility means that isolated food trees, even in urban areas (e.g. see the records in Callala Beach township, in the centre of Figure 2b), may contribute significantly to the viability of remnant populations, as long as food resources are managed effectively.

IMPROVING AND APPLYING THE MODELS

There are several refinements that should be made to the studies we report here. First, as outlined above, we need to improve vegetation classification and resolution, assessment of cover within each vegetation class, and testing of vegetation characteristics (composition and cover) representing good quality habitat for particular species. We expect that this

TABLE 2. Amount and percent of modelled suitable habitat for the Eastern Bristlebird and the Glossy Black Cockatoo that would be affected by two scenarios of fuel-reduction burning.

	Area of suitable habitat (ha)	**Area affected by SFAZ (ha)**		**% affected by SFAZ**	
Simulation		**250 m**	**450 m**	**250 m**	**450 m**
Eastern Bristlebird	90,117	2,150	4,095	2.4	4.5
Glossy Black Cockatoo	30,120	1,343	2,500	4.5	8.3

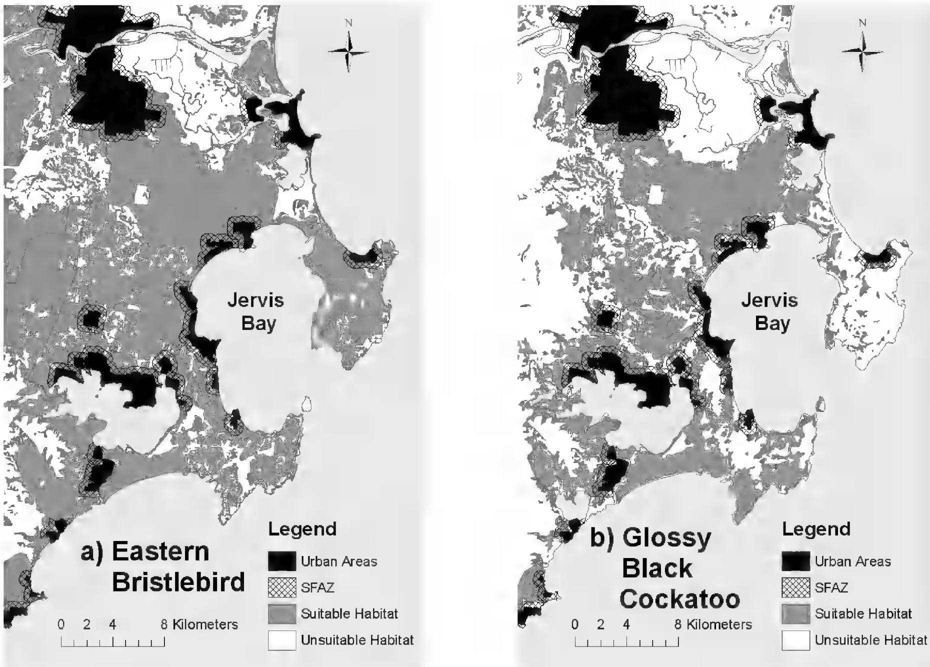

FIG. 1. Maps showing intersection of suitable habitat and Strategic Fire Advantage Zones for (a) Eastern Bristlebird and (b) Glossy Black Cockatoo habitats.

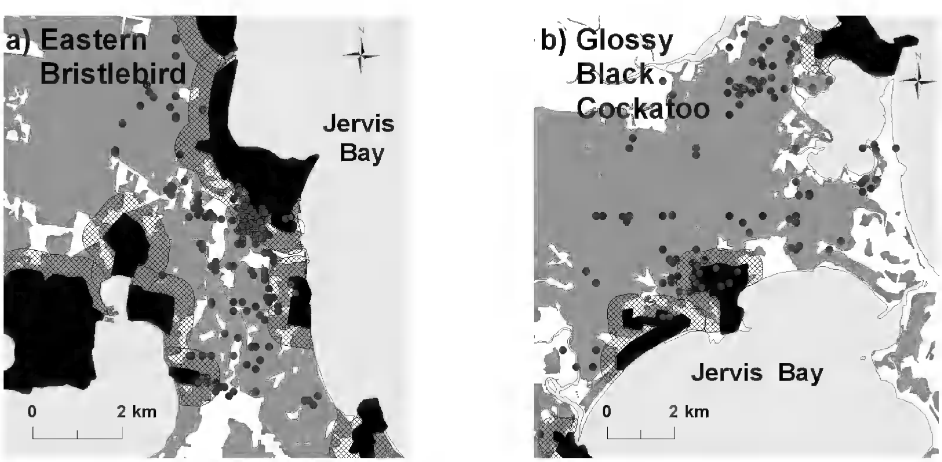

FIG. 2. Local records (dots), urban areas (black), the 450 m SFAZs buffer (cross hatching), and modelled suitable habitat (grey shading) for (a) Eastern Bristlebirds in the Vincentia - Erowal Bay area and (b) Glossy Black Cockatoos in the Callala Bay – Callala Beach – Huskisson area

would result in a significant amount of the landscape that we classified as 'suitable' being reclassified to unsuitable. Second, the buffers we used for the two SFAZ scenarios were generalisations. As the bushfire risk management process is refined, it should be possible to tailor SFAZ widths to particular locations. At the very least, a GIS model could be designed to set the SFAZ buffers based on the intersection of terrain and vegetation layers in particular locations, rather than using a standard width, because slope, aspect and vegetation type are important determinants of the appropriate width of SFAZs.

Third, the relationship between time since last fire and suitability of habitat is poorly known for most threatened species and is generally inferred from estimates of habitat. However, it is possible that, depending on local conditions of soil fertility and rainfall, vegetation in some sites could rapidly return to suitability. As a consequence, the impacts of fuel-reduction activities in the SFAZ may not be so severe. For example, *Allocasuarina* trees burned as part of fuel-reduction activities may recover and produce new cones after only a few years (Collins, 2005) and rapid understorey recovery may make a site suitable for Eastern Bristlebirds soon after a fuel-reduction burn. Thus, further research on the relationship between fire and habitat will allow more precise predictions of the effects of SFAZs.

Even as it stands, the model we have developed will allow those responsible for bushfire management plans to tailor on-the-ground actions to particular circumstances. For example, fuel-reduction activities or SFAZ widths could be modified in particular locations where high quality habitat would be isolated or converted to unsuitable, and scheduling of fuel-reduction activities could be modified according to the impacts on the distribution of high quality habitat of a recent bushfire.

ACKNOWLEDGEMENTS

We thank Dr Marji Puotinen, Dr Laurie Chisholm and Mr John Marthick for their assistance with GIS, and the Department of Environment and Climate Change, Shoalhaven City Council, and the Rural Fire Service for provision of data and other assistance.

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Rob Whelan was Professor of Biological Sciences and Dean of Science at the University of Wollongong until August 2008. His main research interests include fire ecology and conservation biology. He has worked on the effects of aspects of fire regimes on plant and animal populations and on the effects of disruptions to pollinator assemblages on plant mating systems and reproductive success. He is the author of The Ecology of Fire (Cambridge University Press, 1995) and a panel member of the 2004 Council of Australian Governments' National Inquiry into Bushfire Mitigation and Management. He is currently President of the University of Wollongong in Dubai.

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IMPACT OF PRESCRIBED FIRES ON DOWNWIND COMMUNITIES

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REISEN, F. & BROWN S.K. 2008 06 25: Impact Of Prescribed Fires On Downwind Communities. *Proceedings of the Royal Society of Queensland*, 115: 85-91. Brisbane. ISSN 0080-469X.

Prescribed fires are conducted each year across Australia for fuel management and/or ecological reasons. In some regions, during the period of intensive prescribed burning, large quantities of hazardous air contaminants may be emitted and may exceed urban air quality guidelines. The impact on community health will depend on what hazardous pollutants the population is exposed to, the levels of exposures and the potential for such exposures to cause adverse health effects among the community.

The paper will discuss findings from a review of Australian and international literature regarding implications of exposure to bushfire smoke 'air toxics' on community health. Most studies have been on large fires, whether accidental or during forest clearing activities with little to no research on community health and air toxics exposures downwind of prescribed fires. The review has shown that the primary pollutant consistently exceeding air quality guidelines downwind of large bushfires was particulate matter, but it also highlighted that there is clearly a need to further investigate the effects of bushfire on public health. Some of the issues to be addressed include

- Monitoring the seasonal exposure of communities to hazardous pollutants released during prescribed burning activities and determining major factors that influence exposure levels
- Assessing the health implications of such exposures, especially in relation to existing (urban) air quality measures or by developing a bushfire air quality index as a tool for better risk management.

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Bushfires occur regularly across the world and are likely to release substantial quantities of hazardous air pollutants, which has raised concern about potential health impacts on communities. In Australia, the practice is to conduct fuel reduction burns during spring and autumn to remove fine and more flammable fuel and thereby reduce the severity of unplanned fires. During the period of intensive burning activities, bushfire smoke can travel substantial distances to surrounding rural towns, resulting in regular annual exposures to rural communities. The impact on community health will depend on what hazardous pollutants the communities are exposed to, the levels of exposure to these pollutants, and whether adverse health effects are likely to occur in the populations exposed.

While much data are available on urban exposures to hazardous pollutants, there is currently a lack of measurements of these pollutants in rural areas. Rural communities undergo seasonal exposures to toxic air pollutants as a result of prescribed burns, but little is known of the levels of exposure that occur and potential adverse health impacts. This review will present findings on community exposures downwind from major bushfire events, and will consider exposure assessment, health impacts and exposure criteria.

COMMUNITY EXPOSURE STANDARDS AND GUIDELINES FOR URBAN AIR POLLUTANTS

It is well established that urban communities are exposed to a range of air pollutants associated with urban activities, in particular traffic related sources. In Australia, ambient air quality standards are set within the National Environment Protection (Ambient Air Quality) Measure and National Environment Protection (Air Toxics) Measure (NEPMs), as summarised in Table 1 (National Environmental Protection Council, 2003). These NEPMs outline agreed national objectives to protect the health and well being of populations and provide nationally consistent ambient air quality standards for 6 criteria

TABLE 1: National Environmental Protection Measures (NEPMs) for ambient air quality

Air contaminant	Average period	Maximum concentration	Maximum allowable exceedances
Carbon monoxide	8 hours	9.0 ppm	1 day per year
Nitrogen dioxide	1 hour 1 year	0.12 ppm 0.03 ppm	1 day per year None
Photochemical oxidant (as ozone)	1 hour 4 hours	0.10 ppm 0.08 ppm	1 day per year 1 day per year
Sulfur dioxide	1 hour 1 day 1 year	0.2 ppm 0.08 ppm 0.02 ppm	1 day per year 1 day per year None
Lead	1 year	0.5 µg/m^3	None
Particles as PM_{10}	1 day	50 µg/m^3	5 days per year
Particles as $PM_{2.5}$	1 day 1 year	25 µg/m^3 8 µg/m^3	Gather sufficient data for review in 2005
Benzene	1 year	0.003 ppm	Gather sufficient data to develop standard in 2009
Toluene	1 day 1 year	1.0 ppm 0.10 ppm	Gather sufficient data to develop standard in 2009
Xylenes	1 day 1 year	0.25 ppm 0.20 ppm	Gather sufficient data to develop standard in 2009
Formaldehyde	1 day	0.04 ppm	Gather sufficient data to develop standard in 2009
Benzo(a)pyrene	1 year	0.3 ng/m^3	Gather sufficient data to develop standard in 2009

pollutants and a range of air toxics or hazardous air pollutants. The current ambient air NEPMs (24-hour) are 50 µg/m^3 for PM_{10} (particulate matter with aerodynamic diameter less than 10 µm) and 25 µg/m^3 for $PM_{2.5}$ (particulate matter with aerodynamic diameter less than 2.5 µm). It will be shown later in this review that such levels are often greatly exceeded for communities living downwind of large bushfires, though it is unknown whether urban particles and bushfire smoke have equivalent health impacts at these exposures.

COMMUNITY HEALTH IMPACT FROM EXPOSURE TO BUSHFIRE SMOKE

During bushfires, a wide range of air contaminants are released which may cause adverse health effects if communities are exposed to elevated concentrations for extended periods. Air toxics in bushfire smoke are present in both the particle and gaseous phases, and include respirable particles, carbon monoxide (CO), carbon dioxide, nitrogen- and sulphur-based compounds, aldehydes, volatile and semi-volatile organic compounds (VOCs and SVOCs), polycyclic aromatic hydrocarbons (PAHs), dioxins, organic acids, free radicals and ozone (O_3) (Brauer, 1999; Fujiwara *et al.*, 1999; Malilay, 1999; Ward, 1999).

A number of research studies have evaluated the impact of exposure to bushfire smoke on community health. They investigated a range of health outcomes including mortality, increased incidences of asthma exacerbation and respiratory disease, changes in lung function, increase of cardiopulmonary symptoms (coughing, wheezing, shortness of breath, angina), upper respiratory illness, as well as mucous membrane irritation. These studies relied primarily on reviews of hospital-based records (e.g. number of emergency department visits and hospital admissions during bushfire episodes). Very few community-based studies (e.g. epidemiological studies of communities) were conducted other than community surveys. The majority of the studies also focused on particulate matter, which was the primary pollutant that consistently exceeded air quality guidelines. This is a

significant finding, since fine particles in urban air (not from bushfires) have been found to impact the health of urban communities. A summary of the reviewed studies is presented in Table 2.

The studies show several consistent findings:

- Bushfires generated large amounts of particulate matter, especially respirable particles, which greatly exceeded air quality standards. Maximum daily PM_{10} were recorded at approximately 200–1000 µg/m³; any daily PM_{10} above 100–200 µg/m³ was considered a high pollution episode. Other criteria pollutants (O_3, NO_x, SO_2 and CO) may have increased, but generally stayed within the health-based environmental standards. Elevated levels of CO and PAHs were observed in Indonesia during the 1997 fires (Kunii, 1999; Aditama, 2000; Kunii *et al.*, 2002).

TABLE 2. Studies of PM exposures and health impacts on communities downwind of bushfires

	Air Toxics (Units PM [µg/m³])	**Increased health effects**[1]	**Reference**
United States	Max. PM_{10} >150	A, COPD	Duclos *et al.*, 1990
	Not measured	Bronchospastic and irritative reactions	Shusterman *et al.*, 1993
	Not measured	A, B, chest pain	Sorenson *et al.*, 1999
	PM_{10} max. >1000, 16 day >150, 2 day >500	A, COPD and RD	Mott & Meyer, 2000; Mott *et al.*, 2002
	$PM_{2.5}$ max., ave. >65, no increases for PAHs and benzene	RD, H	Ward & Smith, 2001; Bible, 2002
	$PM_{2.5}$ 63; CO 1 ppm	Worsening symptoms in 21 COPD	Sutherland *et al.*, 2005
	$PM_{2.5}$ max. 90–200	Not measured	Sapkota *et al.*, 2005
	PM_{10} max. 215; increases for CO and NO, but not for NO_2 and O_3	Not measured	Phuleria *et al.*, 2005
Australia	Nephelometer readings	A	Churches, 1991
	PM_{10} max. 250, median 18; no difference for O_3 and NO_2	No increase in A or RD	Cooper *et al.*, 1994; Smith *et al.*, 1996
	PM_{10} max. 210	Peak exp. flow rates in children (wheeze), no change	Jalaludin *et al.*, 2000
	Max. $PM_{2.5}$ 70	A esp when $PM_{2.5}$ > 40	Johnston *et al.*, 2002
Southeast Asia	PM_{10} max. 1800; CO very unhealthful; SO_2, NO_2 and O_3 good to moderate	A, pneumonia and worsening of respiratory symptoms	Kunii, 1999; Frankenberg *et al.*, 2002; Kunii *et al.*, 2002
	TSP 3–15 times std; NO_x and SO_2 up to 4 times std; CO up to 15 times std	RI, mortality (resp. failure), cough and RD	Aditama, 2000
	PM_{10} max. 930	A, RI, COPD (>65 yo); elderly and people with A had larger effect	Brauer & Hisham-Hashim, 1998; WHO, 1998; Mott *et al.*, 2005
	High poll. day PM_{10} >210	Death (non-trauma)	Sastry, 2002
	PM_{10} >150 for 15% of year	RI, A; mortality, no change; patients with cough, phlegm, chest tightness; decreased lung function among school children	Hisham-Hashim *et al.*, 1998; Awang *et al.*, 2000
	PM_{10} range 44–60	A (<12 yo) correlated with PM_{10}	Chew *et al.*, 1995
	Not measured	Higher frequency of asthmatic attacks in 28% of respondents	Chia *et al.*, 1995
	PM_{10} doubled to 60–100	A, R; mortality, no change	Emmanuel, 2000
	Monthly PM_{10} increased from 48 to 69; max. daily 218; no change , NO_2 and SO_2	RD, B, COPD, A	Phonboon *et al.*, 1999
	Not measured	Higher freq. for 1–5 yo, >60 yo and outdoor workers	Odihi, 2001
	PM_{10} max. 1000; ave. 110; no change, NO, NO_2 and O_3; SO_2, 4x increase; CO, 10x increase	not measured	Muraleedharan *et al.*, 2000; Radojevic, 2003

[1] Increased medical incidences of asthma (A), respiratory disease (RD), respiratory infection (RI), heart (H), chronic obstructive pulmonary disease (COPD), bronchitis (B) and rhinitis (R)

- Every study reported impact on community health as a result of bushfire smoke exposure with the exception of two studies conducted during the Sydney bushfires in 1994 (Cooper *et al.*, 1994; Smith *et al.*, 1996). The health impacts were most severe for the Southeast Asian fires that involved the highest PM levels and the longest exposure periods.
- Adverse health impacts were found for: mortality rates in some studies; hospital and emergency department visits for asthma or pulmonary disease; asthma exacerbation; reduced lung function; increase of cardiopulmonary symptoms (coughing, wheezing, shortness of breath, heart problems); upper respiratory illness and mucous membrane irritation. Usually the observed health impacts were linked to PM levels on the basis of their high elevation compared to health-based environmental exposure standards.
- Indoor levels of fine particles varied little from outdoor levels downwind of the bushfires (Kunii *et al.*, 2002; Radojevic, 2003; Sapkota *et al.*, 2005), indicating that deposition losses of smoke particles to building surfaces were small and resulted in little improvement of protection from bushfire smoke.

Many studies reported significant health effects at levels of fine particulate matter that greatly exceeded the current Australian one-day NEPM for PM_{10} and $PM_{2.5}$ of 50 and 25 $\mu g/m^3$, respectively. No data are available to confirm the adequacy of these standards for the protection of communities exposed to bushfire smoke. While bushfire smoke and urban particles have similar sizes, there may be significant differences in the constituents attached to the particles and their interaction with other pollutant species present in the smoke, and therefore they may affect community health differently. The studies reviewed above certainly identify similar morbidity impacts for urban particulates and bushfire smoke, and there is limited evidence for similar mortality effects, but the exposure and health impacts are poorly quantified for bushfire smoke. An environmental standard specific to bushfire smoke particles may be more appropriate.

COMMUNITY RESPONSE TO BUSHFIRE AIR TOXICS

There are a range of different options provided by government agencies that help communities to minimize possible health effects during a bushfire smoke event. In general, communities are advised to reduce the amount of physical activity outdoors, to remain indoors with doors and windows shut and avoid any additional indoor air pollution (e.g. burning candles, smoking, using woodstoves, dusting). In some instances, communities may be advised (or required) to evacuate an area, though Australian fire agencies generally recommend home occupants remain under a 'stay or go' plan, where they either leave early or stay if well prepared to protect themselves and their homes. Assuming that home occupants stay in the bushfire smoke, their options to reduce air toxics exposures are:

- Wear a respirator with a suitable protection factor for the levels of air toxics, a strategy which was adopted in Malaysia (Hisham-Hashim *et al.*, 1998). Assuming from this review that fine particles exposures are the primary concern, a particle respirator with a 10-fold protection factor would reduce exposure to one-tenth of the smoke levels. This level of protection has been selected since it is the maximum that can be provided by half-face respirators that are moderately comfortable to wear (Standards Australia and Standards New Zealand, 1994).
- Stay indoors with external doors and windows shut. The basis for this recommendation is that short-term peak smoke levels will be avoided, and that particulate matter is partially deposited to surfaces as it enters a building with ventilation air (a loss we estimate to be ~20%). Recent research in Canadian communities downwind of bushfires determined a median indoor-to-outdoor pollutant concentration ratio of 0.91, indicating that staying indoors did not protect people from smoke exposure (Sapkota *et al.*, 2005).

During the Southeast Asian fires, a Pollutant Standards Index (PSI) from the USA was used to characterise smoke haze events, which takes into account the five criteria pollutants (PM_{10}, CO, NO_2, SO_2 and O_3). PSI levels above 100 indicate that the ambient concentrations exceed at least one of the air quality standards and triggers preventative actions by government, which could include health advice warnings. However the PSI ignores many of the air toxics of Table 1 that may be present in bushfire smoke, limiting the applicability of this index for bushfire haze episodes. A broader index may be appropriate for bushfire smoke.

CONCLUSIONS

More detail of each of these studies is presented elsewhere (Reisen & Brown, 2006) but significant observations on further research needs are presented here. The studies in this review covered major bushfire events, predominantly overseas. They have shown that bushfire smoke consistently caused respirable particle levels in downwind communities to exceed ambient air quality standards, and that this was generally considered to be the primary factor in the adverse health effects observed in communities. However, assigning the health effects to particles assumes bushfire particles and urban air particles have similar health impacts and standards. Exposures to CO, NO_2, SO_2 or O_3 were increased in bushfire smoke, but generally stayed below current standards. Overall, little attention was given to the impact of PAHs, VOCs and aldehydes on community health and to the potential for interactive effects of pollutants.

Some studies observed increased mortality, but most observed increased morbidity, primarily among the susceptible population, which includes children, the elderly, pregnant women, and people with asthma and pre-existing respiratory and/or cardiovascular disease. In general healthy adults quickly recovered from short-term bushfire smoke exposures. An association between bushfire episodes and health impacts, measured by hospital visits for asthma and respiratory illnesses, were observed in South-East Asia and the USA. However, out of three Australian studies, only Johnston *et al.* (2002) reported an adverse health outcome as a result of bushfires around Darwin based on asthma attendances to major hospitals. No explanation is available for this significant difference in findings. There is clearly a need in Australia to further investigate the effects of bushfires on community health.

Even though the studies have all been related to accidental and forest clearing bushfires, it is possible that similar effects of bushfire smoke impact on community health will be observed during a season of intensive prescribed burn activities. In order to determine this potential, monitoring of the air toxics during prescribed burn seasons should be carried out for downwind communities so that the major factors that affect exposure levels are determined. Even in the reports of large fires reviewed in this paper, the determinants of exposure levels have not been sought and these studies have provided little to no information on the fuels combusted in the fires. This could be a key factor in the amount and nature of the pollutants formed, but is largely ignored in discussion. Other important details of the fires were also often lacking, e.g. fire conditions and proximity to communities, as well as meteorological conditions (wind speed, inversion).

Indoor measurements of bushfire air toxics have also been very limited, but generally indicate that staying indoors does not protect people from smoke exposure (other than short-term peaks). There is a need for research on how to protect indoor occupants from bushfire smoke.

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MESSAGE IN A BOTTLE: CULTURE, BUSHFIRE AND COMMUNITY UNDERSTANDING

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Schauble J. 2008 06 25: Message In A Bottle: Culture, Bushfire And Community Understanding. *Proceedings of the Royal Society of Queensland*, 115: 93-99. Brisbane. ISSN 0080-469X.

The communication to the wider community of scientific information about bushfires must be cognisant of a range of pre-existing social and cultural influences that may affect popular understandings. A variety of "cultures" have the potential to influence people's understandings of bushfires and bushfire behaviour. These include the mass media, film, television, literature, art, popular culture and the internet. To illustrate the point, this paper examines the impact of Australian children's literature in perpetuating the notion that discarded bottles and broken glass are a common cause of bushfire ignitions. An analysis of books for children and adolescents since the 19th century reveals acceptance and reiteration of this causation despite its dismissal in scientific and generalist adult texts. The credibility of the notion is reflected in its acceptance and repetition in the adult world, to the point of being quoted in parliamentary debates. The limited academic discourse on the impact of culturally derived understandings of disaster (including bushfires) and community perceptions is discussed.

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Unfortunately, this paper begins with some rather "dodgy" science. But that in a sense is its premise: translating science into practice in the context of bushfire is fraught, not so much with difficulty as with vast differences in and potential barriers to understanding. Facilitating increased community consciousness of bushfires is less of a problem in an era when bushfires are increasingly thrusting into the outer suburbs of Australia's major cities. However, igniting greater community understanding of what it is they are seeing, why this is happening and how Australians should respond to the phenomena is another matter altogether.

Putting bushfire science into the public domain in the hope that it will be understood means competing against a range of established beliefs and some popular misconceptions. Such myths, once established and perpetuated, are difficult to overcome. Exploding houses and fireballs rolling across the landscape remain popular understandings despite the best efforts of fire scientists and community fire educators to dispel them. There are other hurdles, such as the "competing cultures", as they are sometimes called, of science and journalism. There are few better contemporary illustrations of this than the media's handling of the debate over climate change. The news media — and television in particular — are enormously influential in shaping community understandings. In the past, the emergency services have contributed to the mythology in their own communications about bushfire, in the process encouraging a culture of public dependence.

Yet there are many other "cultures" that have also influenced and that continue to influence the way in which physical science is interpreted by a broader audience. This paper uses the example of the notion that bushfires are commonly ignited by discarded bottles and broken glass to demonstrate how basic science, misinterpreted and reiterated, takes on meanings of its own.

DISCUSSION

While bushfires are integral to the Australian landscape and the life and culture of its inhabitants, most will never experience a bushfire first-hand. The same applies to other natural and man-made calamities. This does not diminish the fact that some bushfire events take on broad, even national significance. Stephen Pyne, the influential fire historian, has described the 1983 Ash Wednesday fires as a "cultural epiphany", as significant an event as the fateful Burke and Wills expedition more than a century earlier (Pyne, 1997).

Certainly, the communal artefacts of fire are not necessarily just those of scientific analysis. They are more likely to be creations of popular culture. Large fire events generate enormous media attention.

They also lend themselves to pictorial and literary commemoration. They become events that are in various ways recorded, remembered and imagined: in news reports, film, fiction, poetry, art, photographs, reminiscences, even T-shirts.

But back to the dodgy science. Before the distractions of television, computer games and the internet, almost every school-age child knew the power of the humble magnifying glass. Beyond its practical and legitimate uses, the enterprising also knew that it was possible to use refracted light through this simple instrument to start a fire. All you needed was a glass, some grass or gum leaves and perhaps a bit of paper. It was a simple, practical lesson in physics, though one not always sanctioned by teachers or parents — especially in rural areas where such a combination could have far-reaching consequences unforeseen by a junior fire scientist.

The physics helped fuel one enduring belief about the cause of bushfires. The idea that bottles and broken glass (and even the concave ends of aerosol cans) lying in the bush are often responsible for igniting fires has proved a persistent one. This is despite scientific experiments that found to the contrary, a complete lack of statistical data and even simple common sense. There are roughly 850,000 kilometres of roads in Australia (ABS, 2004). According to the best available estimates, glass bottles make up around 6 per cent by volume of all litter discarded nationally. A recent national litter count conducted on behalf of the Keep Australia Beautiful organisation found that two-thirds of glass bottle litter is discarded on the roadside. On the basis of counts conducted at 151 sites around Australia, the study extrapolated that no less than 320,678 cubic metres of glass lay discarded beside the nation's highways at the time of the count (KAB, 2006).

The first observation that needs to be made is that if glass and bottles on the roadside really were a significant cause of bushfire, then during most summers much of south-eastern Australia and beyond would surely be on fire most of the time. Cheney & Sullivan (1997) suggest that it is theoretically possible that glass bottles or fragments are responsible for the ignition of bushfires. Laboratory tests have shown that a glass bottle containing water can form a lens sufficient to generate combustion by focusing the sun's rays on a fuel bed. However, they note that under field conditions it is "highly unlikely that a bottle or glass fragment will form a lens of the correct focal length and orientation to concentrate sunlight sufficiently to start a fire". They go on to point out that "considering the vast amount of glass and can litter along our roadsides, the chances of ignition by these agents must be very small indeed".

Weber (2000) also acknowledges the theoretical possibility and various attempts to replicate such ignitions. "Glass fragments and discarded cigarettes can cause fires, but this a very unlikely source of accidental ignition," Weber notes, pointing to experiments attempting to evaluate the real possibility of ignition from these sources. Importantly, he also points to mathematical modelling estimating the concentration of heat and its effect on vegetation "showing these to be unlikely ignition sources unless used deliberately". At most, he estimates glass and discarded cigarettes are responsible for just a few per cent of fires. A key 1980s study of the cause and effect of bushfires in Australia does not even mention bottles or broken glass. Nor does a later study of the causes of fire on public lands in Victoria over a 20-year period (Barber, 1986; Davies 1997). In Victoria, an examination of all wildfire, scrub and grassfire reports lodged with the Fire Incident Reporting System (FIRS) between 1 January 2000 and 28 February 2006 found six records of glass or bottles identified as the possible accidental cause of bush or grassfire. In addition, there were two reports of fires started by children using magnifying glasses. On average FIRS records 4000 to 5000 wildfire calls annually, further suggesting glass and bottles are a statistically insignificant fire cause (Harvey, 2006).

Foster (1976) is perhaps the most trenchant critic of the bottle and glass theory, bluntly describing it as "fallacy". After the 1927 Royal Commission of Inquiry on Bushfires in New South Wales listed "glass lying in fields" as one of 10 causes of bushfires, Foster says it became an excuse "used by many graziers to conceal their own culpability for burning off escapes for many years afterwards". He also notes glass appeared as cause in South Australian fire statistics until the 1970s, before concluding that "as a cause of bushfires this phenomenon is discounted by those who have deliberately attempted it". Interestingly, the royal commission's suggestion was being dismissed contemporaneously by the popular author Donald Macdonald in a bushcraft book for boys: "You may by chance get a piece of broken glass lying in such a position that it concentrates the rays of the sun like

a burning glass, but the chance is one in a hundred thousand" (Macdonald, 1930).

Perhaps ironically, given the experimentations of our junior scientists referred to earlier, one of the key repositories of the "truth" that discarded bottles and glass as a bushfire ignition source has been children's literature. Bushfire has had a strong and consistent presence in Australian children's literature since the late 19th century. To begin with, it was just one of a litany of horrors that this alien landscape threw up to challenge colonial new chums. These included floods, Aboriginals, escaped convicts, snakebite and bushrangers (Lees & McIntyre, 1993). Most often fire was portrayed as a destructive enemy, an intruder into the landscape to be beaten back and defeated. Only more recently have other elements — such as regeneration and renewal — found expression in the bushfire story for children (Robinson & Leach, 2002).

Until the 1980s the bush itself was often depicted as a hostile environment. Foster *et al.* (1995) argue that it became almost axiomatic that time in the bush equalled time in danger. This was certainly the 19th century vision of the Australian bush in children's literature. With the growth of fiction for young Australians as a genre in its own right in the mid-20th century, the bush became a place in which the immature were tested and, in some circumstances, made the emotional transition from childhood into young adulthood. Bushfire became a set part of this landscape of danger in many early adventure stories. In 'Harry Treverton: His Tramps and Travels Told by Himself' (1889) by William Henry Timperley the protagonist encounters a bushfire and a ruffian who wants to rob him in the course of one journey (Saxby, 1998). In the popular 'Billabong' series by Mary Grant Bruce, bushfire is one of the consistent threats to station life. In 'A Little Bush Maid' (1913), young Norah demonstrates her pluck when she rescues a flock of sheep in the face of a bushfire (Bruce, 1913).

The highpoint of bushfire as a pivotal theme in juvenile literature was probably the late 1960s and early 1970s. Nimon & Foster (1997) suggest that around this time Australia's unique natural environment and the specific challenges it presented came, in large part, to define the "Australian-ness" of stories for children and adolescents. Landscape came to signify nation. Yet post-war Australia was, of course, simultaneously a period of rapid urban expansion. The landscape of bush and bushfires was for many adolescents increasingly remote: "Rural Australia in its most terrifying manifestations furnished the settings for the majority of the early adolescent novels written in this country. Merely to live here appeared a potentially dangerous enterprise, especially for those approaching puberty."

Rather than merely being part of the backdrop of danger in the bush, bushfire itself became central to the story. Indeed, bushfire stories seemed to break out all over the place. This was coincidentally a period of significant bushfire activity across Australia. Several major bushfires affected rural and outer suburban areas in the south-eastern states and also Western Australia. In Victoria, the Dandenong Ranges (1962, 1968), the Yarra Valley (1962), Longwood (1965), Lara (1969) and central Victoria (1969) experienced large bushfire outbreaks which claimed a total of 37 lives and more than 600 homes, together with extensive stock and property losses. Tasmania endured the devastation of the 1967 fires, which killed 61 people and destroyed 1700 homes. In New South Wales, fires claimed several lives around Goulburn in 1965 and there were other major outbreaks on the South Coast in 1968. In Western Australia, fires destroyed the town of Dwellingup and other communities in the south-west in 1961 (Esplin *et al.*, 2003; Luke & McArthur, 1978).

These real fires undoubtedly helped spark several fictional outbreaks. One of the best known is Ivan Southall's 'Ash Road' (1965), winner of the Children's Book Council Book of the Year in 1966. Other titles to emerge at this time include 'February Dragon' (1965) by Colin Thiele, 'Bushfire' (1967) by Alan Aldous, 'Wildfire' (1973) by Mavis Thorpe Clark, 'The Bush Bandits' (1966) by Betty Roland, 'The Ring of the Axe' (1968) by James Preston and 'Family at The Lookout' (1972) by Noreen Shelley. The fires also prompted two quite different but potent picture books — 'Ash Tuesday' (1968) written by Joan Woodberry and illustrated by Max Angus and 'The Death of a Wombat' (1972) written by Ivan Smith and illustrated by Clifton Pugh.

Without over-estimating the importance of these works, it is arguable that they influenced the perceptions of a generation or two of Australians when it came to bushfires. Earlier generations had been exposed to a suite of other books, many of them written in Britain for British children about events in Australia that their authors had never themselves witnessed. Other

Australian authors for children, such as Mary Grant Bruce, had written about fire, but never before had the topic been covered in such detail. Later authors similarly picked up and used bushfire as a theme, again often in response to particular fire events.

The 1983 Ash Wednesday fires in South Australia and Victoria inspired a fresh round of juvenile bushfire literature. These fires killed 75 people, burned some 418,000 hectares and destroyed 2400 homes along with significant stock and other property losses (EMA, 2006). Among the Ash Wednesday books is one of the least forgiving juvenile novels about bushfire: 'Firestorm!' (1985) by Roger Vaughan Carr. The author and his family lived at Aireys Inlet on Victoria's south-west coast and lost their own home in the fires. So too did Marguerite Hann Syme, author of the novel 'Burnt Out' (2001) and a picture book 'Bushfire' (2000). Another work was 'Fire on the Ridge' (1989) by John Wells, while Colin Thiele was prompted by Ash Wednesday to return to bushfire in 'Jodie's Journey' (Thiele, 1988).

The influence of fires in and around Sydney during the 1990s and 2000s and more recently in the Australian Alps has been comparatively slight, although bushfire has again found favour as a theme in a range of juvenile literature (Croasdale, 1998; Pausacker, 1998; Harris, 1999; Kelleher, 2000; D'Ath, 2005).

There is, in sum, a substantial body of children's literature dealing directly with bushfire. Over the past century, some 30 juvenile novels have been published which involve bushfire as a major theme, plus another 10 illustrated story books. In addition, there is a range of other educational material published for children over and above any official fire service or government publications. This body of work represents a significant influence on generations of young Australian minds, which for the past half century have also had the influences of film, television and more recently the internet providing even more understandings and interpretations.

What children have been reading about bushfire includes, among some quite accurate and often gripping depictions, is also some pretty dodgy science. For it is in this realm that the discarded bottle not merely survives, but prospers as a source of bushfire ignition. In one of the most poignant and most popular of illustrated Australian stories, 'The Death of a Wombat' (Smith & Pugh, 1972), the glass that starts the fire that kills the hapless *Vombatus ursinus* is not even clear, but oddly a discarded, unbroken, brown bottle.

In 'The Bush Bandits' (Roland, 1966) the guilty bottle appears along with another unlikely ignition source ("two dead branches rubbing together could generate a spark, or a broken bottle heated by the sun"). Thankfully, in 'Ash Road', "no one dropped broken glass or bottles that might by chance concentrate the sun's rays onto a flammable substance". In 'February Dragon', the "empty bottle" appears in a list of possible bushfire causes. Mr Pine tells his children that almost all fires are started by humans and "almost always through carelessness":

"Campfires, broken exhaust pipes, bad spark arresters on tractors and railway trains, magnifying glasses and empty bottles, hot ashes, incinerators, welding gear, lamps, electrical faults. But most of all from silly people with cigarettes and matches."

'Wildfire' (1973) by Mavis Thorpe Clark also provides a list of causes: campers not putting out campfires, motorists tossing cigarette butts from the window, graziers burning off or using vermin baits carelessly. She echoes the curious observation of the 1939 Stretton royal commission in Victoria that "sometimes lightning started a fire but not often; and then the blaze was generally followed by rain that extinguished it". The physical science and past experience indicates otherwise, with around 25 per cent of bushfires in Victoria started by lightning. The 2002-03 Alpine fires were all started by lightning, burning for several weeks before rain extinguished them (Davies, 1997; Esplin *et al.,* 2003).

In 'Eleanor, Elizabeth' (1984) by Libby Gleeson, the bottle is back in the frame — indeed, it is the sole cause of bushfires mentioned in the text. Eleanor's mother carefully packs drinks for a picnic on a high fire danger day. "I've emptied the drinks into plastic bottles," she tells her daughter. "Glass is too much of a fire hazard in this weather". The first suspicions about fires breaking out on the property in 'Black Earth' (Forrestal, 2004) point to arson. To begin with one of the children playing with matches is suspected, and then a mysterious man is seen camping in the bush. In the end the cause turns out to be the hot exhaust of a new quad bike being used on the vineyard. Refracting glass is also mentioned here, but this time it is the more plausible proposition that a child may have been starting fires by playing with a magnifying glass.

Aside from fiction, broken glass and bottles have also occasionally found their way into educational texts. 'Bushfires' (1979), one of the 'Australian Fact Finders' series for young readers written by Michael Dugan, gives the broken bottle as a source of ignition further credibility: "Broken glass or bottles left lying in the bush can also lead to bushfires. A piece of broken glass can act as a magnifying glass. It concentrates the sun's rays on dry glass or twigs and sets them alight." He repeats and expands upon the theme in 'Bushfires' (1996), one of the 'Australian Disasters' series of educational books for young people, again stating that "glass can act as a magnifier".

While the mainstream scientific and non-scientific literature of bushfire has largely discarded it, the continuing reiteration of the glass bottle theorem in children's literature has ironically ensured its survival in the adult world. It is especially prevalent in discussions about roadside litter. During debate on the ACT Bushfire (Amendment) Bill 1998, the Minister for Urban Services, Mr Smyth, told the Legislative Assembly that fire "can commence from light focused through a broken Coke bottle" (ACT Legislative Assembly, Hansard, Week 10, 26 November 1998). One rural MP confidently told the NSW Parliament during a debate on changes to the littering laws in 2000 that "in summer time bushfires are started by the sun burning through glass bottles thrown from car windows" (NSW Legislative Council, Hansard, 4 May 2000).

In Western Australia, the Shire of Roebourne's code of ethics for travellers warns visitors to the Pilbara to "be alert to prevent causing bushfires", the listed causes of which are "electrical faults, cigarette butts, broken glass and even vehicle exhausts" (Roebourne, 2001). In a 2003 discussion paper on litter abatement, the Keep Australia Beautiful Council (WA) points to "broken glass igniting dry vegetation" as a threat to the state's biodiversity (KABC, 2003). The travel website walkabout.com, published by Fairfax Digital, earnestly warns visitors to Australia to "respect fire bans (broadcast on the radio) and be careful with cigarette butts and broken glass which can ignite bushfires in hot, dry weather" (Fairfax, 2006).

Why has the myth of the broken glass or the abandoned bottle been so persistent? There is probably some merit in Foster's (1976) observation about rural landowners looking for a convenient scapegoat. But there may also be another reason, at least in relation to the persistence of the bottle in children's literature. An accidental cause is a convenient way of shrouding, from children at least, the less palatable fact that many of the worst Australian bushfires in recent years have been the work of arsonists, sometimes acting with dark and malicious intent.

CONCLUSIONS

There has been remarkably little study of the impact of broader cultural influences on community understandings of fire and other disasters. In the United States, Quarantelli (1980) led the way with an examination of disaster movies and the impact these had upon community understandings. More work has been done on the impact of the news media on public perceptions and some is now being specifically undertaken in conjunction with the Bushfire CRC in relation to the media and bushfires in Australia (Cohen *et al.*, 2006).

For those trying to bring meaning through science to a broader public in the hope that their responses to bushfire will be grounded in better understandings of such events, there are some considerations in this context. Australians have had a century to "unlearn" about the bush. In 1906, 52 per cent of Australians lived in towns and cities. A century later, that figure is around 90 per cent. Their understandings of what goes on in this alien, fire-rich landscape into which cities are sending tentacles and into which more and more Australians are making "sea change" or "tree change" escapes is equally affected by a range of extraneous cultural influences.

These include such seemingly innocuous sources as film, fiction and children's books. Wachtendorf (1999) argues that all cultural texts, no matter what their form, are relevant to our knowledge of disasters and offer researchers "clues" as to how different groups experience crises. As Liverman & Sherman (1985) have pointed out, "disaster novels often convey a sense of scientific accuracy". At the same time, films and novels give succour to many myths. In the Superman movie, the earth opens up in gaping holes when the earthquake hits. The post-disaster social behaviours such as looting, panic, outbreaks of crime and disorder remain common themes in disaster fiction despite having long been debunked by researchers (Quarantelli & Dynes, 1972).

Pyne (2004) has recently pointed to a disconnection between America's vernacular fire reality and its high

culture. He argues that wildfire in America cries out for philosophy, history, ethics' literature, economics and political theory. Instead, the language of fire has become the jargon of the technical manager and sensational journalism: "a subject that goes to the heart of our identity as a species ends up as government reports, bowdlerised war stories, or a genre of juvenile sports literature". Fire, in order to be fully understood by the wider community, needs to engage the interest of the intelligentsia, he argues. Despite the influential role of authors such as Normal Maclean and, indeed, Pyne himself, what fire needs, he concludes, is a poet.

The message is clear enough for those seeking to translate science into practice. In Australia, popular culture — even the seemingly innocent children's book — is a rich source of information that those concerned with perception and response to bushfire cannot afford to ignore. Evidently, that information varies in quality. Yet it is reasonable to assume that people will retain and interpret at least some of this information as an accurate reflection of reality, depending on its source and just how convincingly it is delivered. That is the message in the bottle.

ACKNOWLEDGEMENTS

My thanks to Associate Professor Don Garden, School of Social and Environmental Enquiry, University of Melbourne, and Professor Emeritus Joe Scanlon, Carleton University, Ottawa, for their comments and suggestions regarding this paper.

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FIRE INTERVAL SEQUENCES TO AID IN SITE SELECTION FOR BIODIVERSITY STUDIES: MAPPING THE FIRE REGIME

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Wittkuhn, R.S., Hamilton T. & McCaw L. 2008 06 25: Fire Interval Sequences To Aid In Site Selection For Biodiversity Studies: Mapping The Fire Regime. *Proceedings of the Royal Society of Queensland*, 115: 101-111. Brisbane. ISSN 0080-469X.

Determining the impact of fire regimes on biota is often limited by the lack of good knowledge about fire history: where have fires occurred, how big were they, and at what time of the year and with what intensity did they burn? On the other hand, where fire history has been well documented, the complexity of this information can be daunting. In this paper, we show how a simplification of complex fire history data into sequences of fire intervals can provide a basis for studying the impact of contemporary fire history on biodiversity. We used a retrospective approach to classify historical fire intervals into descriptive units (short, moderate and long intervals). In particular, we wanted to view the sequence of past fire intervals within the spatial framework in which it exists (i.e. where in the landscape do contrasting fire interval patterns exist?). Our study centred on an area of 50 000 ha northeast of Walpole, Western Australia, that was last burnt in the fire season of 2002/03. This provided a unique opportunity to retrospectively investigate fire interval sequences using a common, recent and widespread fire as a baseline. Our fire history dataset spanned back to 1972, providing a maximum of 30 years between the least- and most-recent fires. All fire occurrence data was held in a Geographic Information System (GIS), with spatial information for fire boundaries, and years in which fires occurred. By exporting the database file (.dbf) associated with fire occurrence into Excel, we were able to assign codes for short, moderate and long intervals between successive fire events. Then, these codes could be combined for each landscape patch with a unique fire history in space and time to produce sequences of fire intervals. The result of this technique was a map of polygons with a display of their fire interval sequence in reverse time series. This representation of the results provided a quick overview of how the pattern of fire intervals differed across the study area, and we found it useful in determining those areas that have burnt with either successive short or long fire intervals. This method is innovative, cost-effective and attempts to deal with the problems of complex multidimensional data.

❐Geographic Information Systems, fire history, fire management

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Research on the influence of fire on biodiversity often suffers from a lack of underlying knowledge of fire history for the study area. Particularly for contemporary periods (say, the last 50 years), the underlying fire history may differ significantly between study sites, and this may influence the species composition even before experimental treatments are applied. In most cases, fire history is not known or is poorly recorded with inaccurate spatial resolution (Kitchin & Reid, 1999). In rare cases, fire history information is accurate in space and time, and provides an excellent background to ecological studies (e.g. van Wilgen *et al.*, 2000; McCaw *et al.*, 2005). However, with accurate information comes complexity in both space and time, making the identification of replicate study areas that have had the same fire history problematic. Typically, a statistical mean or median fire interval has been applied to the landscape when questions of fire regime are raised (Everett *et al.*, 2000; Wright & Agee, 2004), and there are still uncertainties in applying

statistical distributions to fire interval data (Baker, 1989; Gill & McCarthy, 1998). However, average fire intervals and their variation at a landscape scale may not provide enough detailed information to explain differences that are observed in biological systems at a finer scale.

There is increasing evidence from around the world that inappropriate fire regimes are having a deleterious effect on ecosystems. In some environments, evidence suggests an increase in fire frequency relative to historical frequencies, resulting in either a loss of species or changes to the community composition (York, 1999; Bradstock *et al.*, 1997; Kodandapani *et al.*, 2004; Russell-Smith *et al.*, 2004). In other environments, fire intervals have become longer than those experienced historically, which can result in equally dramatic consequences for biodiversity (Abrams *et al.*, 1995; Heyerdahl *et al.*, 2006), especially when coupled to changes in forest structure and abnormally high fuel loads which result in high intensity fires (Wright & Agee, 2004; Stephens & Fulé, 2005). In southwestern Australia, fire has been used since the 1920's as a management tool for strategic fire suppression (protection of life and property), and conservation of biodiversity (McCaw & Burrows, 1989). This is probably one of the longest periods in the world (post-European colonisation) involving the active use of prescribed fire from a strategic land management point of view. Since the 1930's, maps were developed showing areas that were burnt in prescribed fires, or that burnt as a result of unplanned fires. Since 1953, there is almost a complete record of all fires that occurred across the southwest of the state, though the spatial accuracy of these is low in some cases as a result of base map quality (i.e. lack of cadastre information or absence of roads etc.). This historical fire area information has recently been captured into a Geographical Information System (GIS) (Hamilton *et al.*, in prep.), resulting in a fire history database (FHD) that contains the temporal and spatial fire history information for the Warren Region (one management region within the southwest of Western Australia; Fig. 1).

The work described in this paper forms part of a larger study to investigate the effects of contemporary fire history on the diversity and abundance of vascular flora, vertebrate and invertebrate fauna, fungi, mosses and lichens. The FHD for the Warren Region was our source of information to determine historical fire regimes, and consequently, where the study plots should be located. The study constitutes a descriptive (i.e. non-manipulative) research project to investigate the effects of repeated fire on the landscape biota. Of particular interest are those organisms that are 'fire-regime sensitive'; that is, they may become locally extinct as a result of repeated fire that is of an inappropriate frequency, season, intensity or combination of such (Gill & Bradstock, 1995; Burrows & Friend, 1998). This includes organisms that are sensitive to fire that is too frequent, or those that are sensitive to prolonged exclusion of fire (Keith, 1996; Menges *et al.*, 2006). In the latter case, wildfires can burn with unusual severity leading to long-term changes in vegetation structure and damage to soil properties, water cycles and nutrient cycling (DeBano *et al.*, 1998).

Morrison *et al.* (1995) and Cary and Morrison (1995) have shown that the sequence of fire intervals (particularly in relation to short fire intervals) has important implications to community structure and species composition. In particular, they demonstrated that shorter fire intervals were associated with a reduction in the number of species present in the community, and that repetition of fire intervals 1–5 years long was associated with an increase in the abundance of herbaceous fire-tolerant species. They demonstrated that the sequence of fire intervals may be important to the abundance and diversity of flora in sclerophyllous sandstone communities near Sydney. In contrast, there are few studies that investigate the effects of longer fire intervals on biota in Australian communities.

Our study aims to investigate the effects of both short and long fire intervals on biota in southwestern Australia, concentrating on the pattern or sequence of fire intervals. Because we are undertaking a retrospective study, the communities to be studied are very much determined by what pattern the fire history shows across the landscape, and this paper provides a description of how we have characterised fire intervals in our chosen study area. The biological surveys are currently underway, and will be published upon completion. The current paper is a purely methodological paper. We describe a process that simplifies patterns of fire intervals through time for discrete 'patches', classifying each interval into short, moderate or long, and joining these classifications together to show the sequence of fire intervals through time for each 'patch' of land. This process takes place in a GIS and through export of database

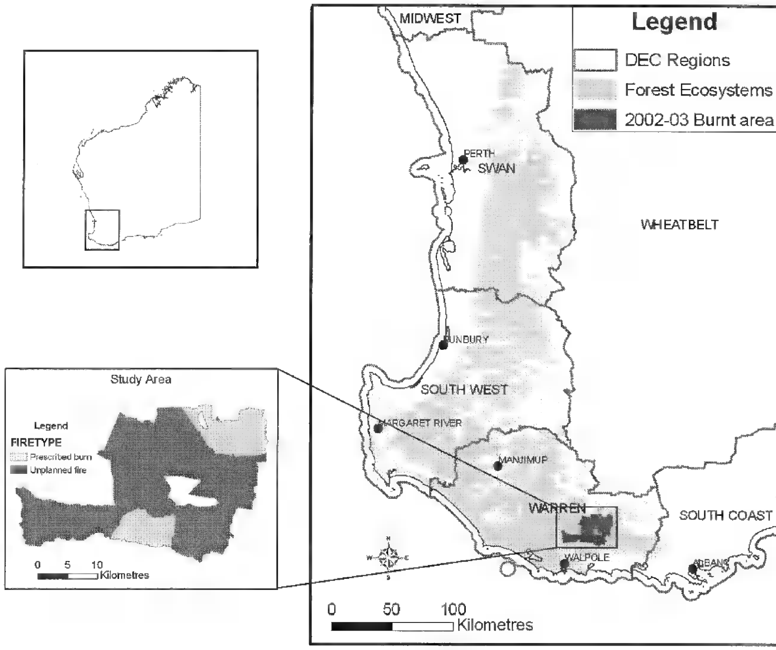

FIG. 1: Study area for the purposes of mapping fire interval sequences. Sequences were developed for a ~ 50 000 ha study area within the Warren Region, for which a complete fire history database has been developed (1953–2006). The study area was all burnt in either prescribed burns of spring 2002 or unplanned burns over summer (December – March) 2002/03.

files to Excel, and the sequences of fire intervals can be mapped in the GIS to show the historical fire interval patterns across the landscape to aid in site selection for ecological studies. The main aim was to simplify the GIS output by showing temporal information in a spatial context, without losing the detail in the historical information.

MATERIALS AND METHODS

THE FIRE HISTORY DATABASE (FHD)

Since 1937, forest fires in southwestern Australia have been recorded on paper maps by relevant districts of the Forests Department (prior to 1985), Department of Conservation and Land Management (1985-2006) and Department of Environment and Conservation (2006-present). Old maps (prior to 1995) were photographed to microfiche to economise on space, and recently have been scanned and digitised into a GIS for the Warren Region (Hamilton *et al.*, in prep.). More recent fire data (1995–present) were contained in raster or vector format and was incorporated into the FHD. For the purposes of our study, fire history information prior to 1972 was not included for determination of fire intervals as the accuracy of some fire boundaries was questionable due to poor cadastral information or other resources at the time (such as aerial photography) to aid accurate mapping. Fire data for each year was digitised in ArcView™ 3.1 as polygon shapefiles, and attributed information that was contained on the maps or in the database, such as whether the fire were a prescribed burn or unplanned fire (wildfire) and in which season it burnt. The data were validated by investigating complementary fire records (including aerial photographs), speaking with fire officers in the districts and re-evaluation of the original images. To complete the FHD, layers from all years were merged so that it was possible to determine all fires that occurred in a given area over time. A detailed description of the methods used to create the FHD is given in Hamilton *et al.* (in prep.).

STUDY AREA

In order to investigate the effect of past fire regimes on biodiversity, it was necessary to remove or minimise the effect of other variables that may also affect biodiversity across the landscape. The most obvious of these is time-since-fire (Morrison *et al.*, 1995). In jarrah forest communities, flora species richness and diversity increases with time-since-fire until 3–5 years, after which a gradual decline occurs (Bell & Koch, 1980). The initial increase results from species that are 'fire weeds' (germinate from soil-stored seed after fire and complete their life cycle within 5–7 years). Similarly, invertebrate communities in jarrah forest undergo changes in composition and abundance within the first four years after fire, by which time higher taxa return to pre-burn levels, while some early taxa that prefer disturbed habitats still remain (van Heurck *et al.*, 1998). We considered it important to have all study sites at the same time-since-fire to account for this, and preferred sites that were burnt 3–5 years previously so that the number of extant plant and invertebrate species was maximised (Bell & Koch, 1980; van Heurck *et al.*, 1998).

We identified a contiguous 50 000 ha study area 30 km north-east of Walpole, all of which was burnt in either unplanned fires or prescribed burns in the fire season of 2002/03 (Fig. 1). This area also contained a range of contemporary fire regimes within common vegetation complexes, and was relatively free of disturbances such as logging. The vegetation in the study area was a mosaic of predominantly jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*) forest interspersed by winter-wet shrublands that varied in species composition, and ranged from low woodlands to shrublands and sedgelands.

The climate of the area is Mediterranean, with mild, wet winters and warm, dry summers. Shrubland areas become inundated over winter and into spring and have shallow soils underlain by laterite. Over summer these areas dry out, and become highly flammable. The landscape is undulating, with the jarrah forest communities on the hills and the shrublands in the valleys. Lightning strikes are common in this area when summer troughs form down the west coast of the state, and hence unplanned fires form a significant part of the fire history of the area.

CLASSIFYING FIRE INTERVALS

For our overall study, we were interested in the sequence of fire intervals (temporal pattern) across the landscape (spatial pattern), so that we could identify sites with a similar vegetation type that had contrasting fire histories. We were also interested in finding replicate vegetative communities with similar fire histories, using vegetation complex information that had already been developed by Mattiske & Havel (1998). For the purposes of this paper, we will describe only the methodology involved in classifying the fire intervals (i.e. the temporal pattern). The spatial pattern of fire occurrence is maintained by the polygons in the FHD (vector data); hence patches created by overlapping fires are also maintained using this method. This was an essential component of this work, as we were using this method to identify potential study areas for plot establishment. The methodology of overlaying fire interval sequences and vegetation complexes will be dealt with in a later paper.

Landscape classification methods include the aggregation of spatially explicit information using grid-based data (raster overlays in GIS; e.g. del Barrio *et al.*, 1996; Moritz & Davis, 1997; Bollinger & Mladenoff, 2005); combining spatial databases with data types held in non-spatial databases (e.g. Le Maitre *et al.*, 1993; Richardson *et al.*, 1994); image classification methods (particularly from remote sensing data; e.g. Lentile *et al.*, 2006); and GIS toolbox methods such as the use of intersection, union or ESRI™ GIS extensions (Johnston, 1998). While these methods have particular applications, we were searching for a novel way of mapping the sequence of temporal data; not so much as a way of classifying fire regimes, but for use as an exploratory tool into the sequence of fire interval data. Mean fire interval, and the variance around the mean, has commonly been used to explain differences between sites (e.g. Gill & McCarthy, 1998; Groven & Niklasson, 2005). However, we were interested in the sequence of fire intervals back through time, and how these differed between sites within our study area. We considered the measurement of variation around the mean fire interval as being too coarse for the purposes of our study.

The intention of this work was to maintain all of the fire history information in a GIS database, but to add an extra column to the attribute table that provided a simplification of fire interval data. The original format of the data was that each polygon in the FHD was a result of overlapping fires, with information on years burnt. Here, we describe a method we used to calculate actual fire intervals, then classify these intervals into

three groups (short, moderate and long), then join each of the classifications together into a sequence for a given polygon.

The interval between fires was inferred by interrogating the overlapping polygons (fire events) for any location. The process followed to create fire interval sequences is detailed in the following section. Since the aim of our study was to investigate the influence of fire history on biota, we reasoned that the fire intervals could be classed as 'short', 'moderate' or 'long' based on attributes of the flora and local knowledge on the minimum interval that could carry a fire and a fuel age that would support a highly intense fire under normal (non-extreme) summer conditions. The species that are most at risk of depletion from a frequently burnt landscape are those that have a long primary juvenile period, are serotinous (retain seeds in the canopy) and are killed when subject to 100% scorch (Whelan, 1995). Less information is available on how long species live for, and thus how to classify a long fire interval. In our case, we made a classification based on what fuel age a potentially large, intense fire may occur for the study area. Hence, our classifications of short, moderate and long are not solely based on vital attributes of known flora, but also on the physical nature of the fuels and potential for large, intense fires.

CREATING FIRE INTERVAL SEQUENCES

A unioned FHD layer was created using ArcMap™ 9.1 so that only one polygon existed for each location with a unique fire history. A unique identifier (id1) was created for each location. Using the presence and absence of attributes for each year, a 'Fire Frequency' (number of fires for a given location since 1972) was also created. The merged FHD was then unioned with this dataset, with the effect of adding back multiple layers for each location and associated attributes. A second unique identifier (id2) was created for every overlapping polygon. This associated table from the shapefile was used as the basis for calculating the related fire interval sequences. This table was converted to a Microsoft Excel format.

Using Excel, all records were sorted firstly by the unique identifier (id1), and then by year of fire in a descending order. This had the effect of grouping the data into spatial groups with the records for each group being arranged from most recent to least recent. Each spatial group of data represented the overlapped fire history data contained within one polygon. Because our cut-off point for fire records was 1972, there existed some intervals that represented the time from the least-recent fire back to 1972 (when a fire may or may not have occurred for the given polygon). Where a fire did not occur in 1972, this data is said to be left-censored (Polakow & Dunne, 1999); hence this incomplete fire interval was ignored for the purposes of our study. The most recent fire in 2002/03 was common to all polygons, hence our data contained no right-censored data (Polakow & Dunne, 1999). Fields were calculated for each fire interval (number of years between fires). For the study area the maximum number of intervals was four (resulting from a maximum of five fires over 30 years), but the methodology would be applicable to any scenario with ≥ 1 fire interval(s). For each record, fire intervals were calculated as the difference between the year of the fire for that record and the year of the fire for the previous record (a pair). Hence, in the GIS attribute table, the length of the inter-fire period between a pair of records appears in the same row as the most recent fire of that pair. There is no interval associated with the last record as it is the older year of the final pair. All calculated fire intervals are therefore real intervals (in years), and the sequences we describe in the following paragraph are derived directly from these real intervals.

The next step was to simplify the fire interval data into classifications for mapping purposes. We used a simple index of short (≤ 5 y), moderate (6–9 y) and long (≥ 10 y) intervals. In the Excel spreadsheet, the field 'interval type' was created with reference to the fire interval (in years) for each record, and assigned either a number '1' (if fire interval ≤ 5 y), '2' (if interval 6–9 y) or '3' (if interval ≥ 10 y). Another field, 'interval pattern' was created for the first record in each spatial group, by multiplying the first 'interval type' by 1000, the second by 100, the third by 10 and the fourth by 1, and adding together the results. This produced a four-digit whole number that has the effect of arranging the four interval type classes assigned to each fire interval into a sequence from most-recent to least-recent. For example, if the actual fire intervals going back in time are 11 years ('interval type' 3), 3 years ('interval type' 1), 6 years ('interval type' 2), 10 years ('interval type' 3), then the fire interval sequence that results is '3123', and exists in a single GIS data column that can be displayed in the GIS interface. The process of creating fire interval sequences for given polygons is shown diagrammatically in Figure 2.

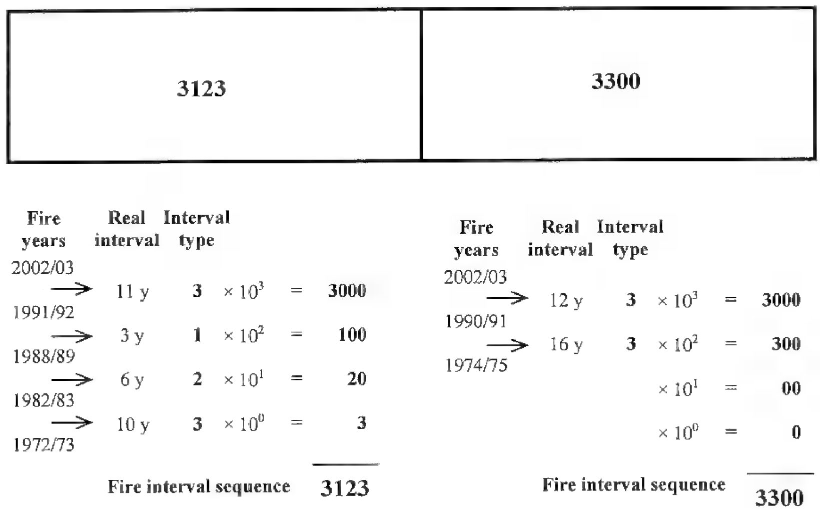

FIG. 2: Example for the calculation of fire interval sequences for polygons contained in the GIS fire history database. For the two polygons, actual fire years are shown. From this, the real fire intervals are calculated as the difference between subsequent fire years. These are then classified into interval types, either as short (≤ 5 years, designated by '1'), moderate (6–9 years, designated by '2') or long (≥ 10 years, designated by '3'). By multiplying the first interval type by 1000, the second by 100, third by 10 and fourth by 1, then adding all together, these interval classes are combined into a single whole number (shown inside the polygon boundary). This 'fire interval sequence' is contained in its own column in the fire history database, and can be used to map temporal information across the landscape. Where a fire interval sequence contains a '0' (always at the end of the sequence) this indicates that fewer than the maximum number of fire intervals for the entire dataset are contained in this polygon (as occurs in the polygon on the right).

Using the unique identifier (id2) for each record, the calculated fire interval data (consisting of actual fire intervals in years, and the fire interval sequence) was re-joined to the fire history shapefile (containing information on fires for all years). This allowed for a spatial representation of the interval sequence, and allows the user to quickly map the fire interval sequences and then obtain more accurate information on all the fires from the attribute table in the GIS database.

APPLICATION OF THE FIRE INTERVAL SEQUENCE – SITE SELECTION FOR BIODIVERSITY STUDIES

Areas that have experienced successive short fire intervals (two intervals ≤ 5 y; designated by a code of '11') or successive long fire intervals (two intervals ≥ 10 y; designated by a code of '33') at some stage in the last 30 years were considered biologically important and were highlighted as potential study areas (Fig. 3b). Areas that have been only 'twice burnt' since 1972 (having only one fire interval) were categorised separately. 'Mixed' fire intervals were defined as all other areas without successive short or long fire intervals that have been burnt more than twice, i.e. contain at least two intervals. This class may contain significant variation within interval sequences. Successive short and long fire intervals and mixed intervals were mapped for all locations within the study area (Fig. 3b). The rationale behind this is that successive short fire intervals or successive long fire intervals may result in localised extinction of 'fire regime-specific' taxa, but those with a mixed disturbance frequency (the 'Intermediate Disturbance Hypothesis'), will maintain a higher diversity of species (Whelan, 1995; Huston, 2003). This postulation is supported by Morrison *et al.* (1995) who presented quantitative data that suggests more variability in the inter-fire interval through time will result in increased species richness at a point in the landscape.

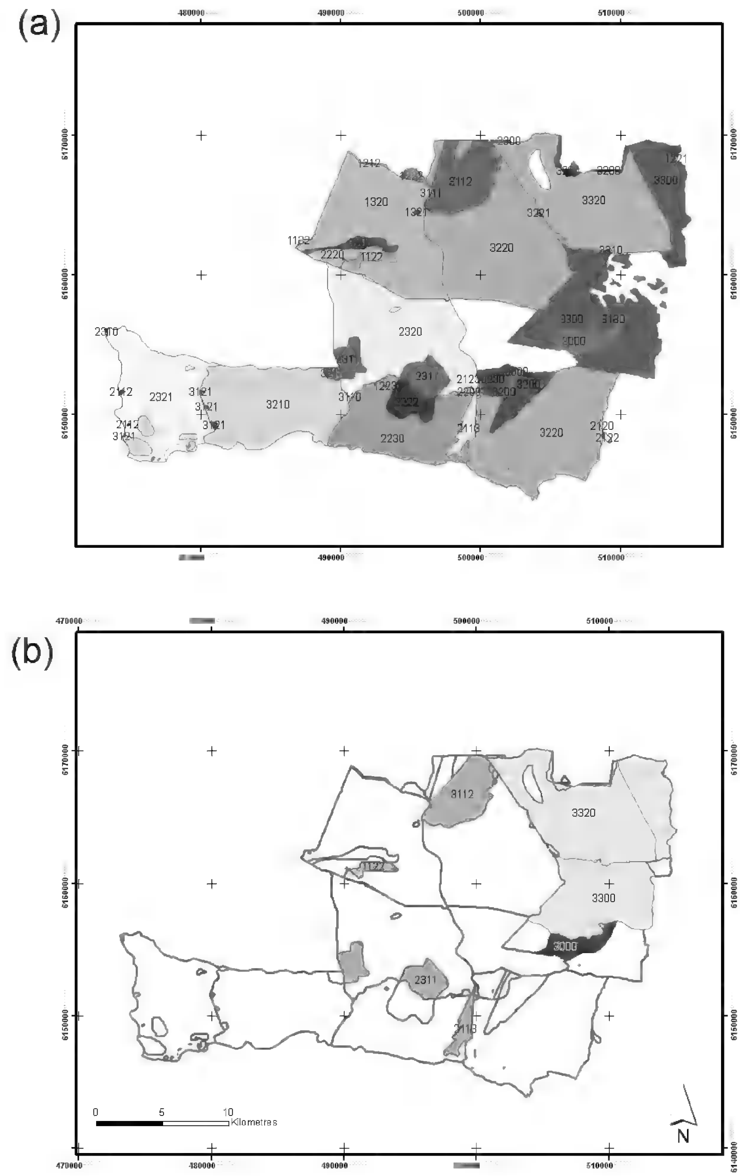

FIG. 3: Depiction of fire interval sequences in reverse time series for our study area 30 km northeast of Walpole. (a) Fire interval sequences for the study area, where '1' = fire interval ≤ 5 y; '2' = fire interval 6–9 y; 3 = fire interval ≥ 10 y, as described in the text and in Fig. 2. Areas with the same shade have the same fire interval sequence. Using colour shading in the GIS system provides a rapid graphical representation of fire interval sequences across the landscape. (b) Fire interval sequences containing '11' (successive short intervals; medium grey shading) or '33' (successive long intervals; light grey shading) mapped against mixed intervals (no shading) and those areas burnt only twice since 1972 (fire interval sequence of '3000'; dark grey shading).

RESULTS AND DISCUSSION

Converting complex fire interval data into a single number sequence that represents the pattern of fire intervals through time allows the user of the FHD to rapidly assess the spatial patterning of historical fire intervals across the landscape. The output that is generated simplifies the complex temporal and spatial components, but retains detailed information on individual fire events, including type (prescribed or unplanned), year and season of the fire. This is demonstrated in Figure 3a that shows all fire interval sequences in our study area. The intention of this methodology was to be able to show temporal data in a spatial context as a basis for then investigating the more complex dataset.

For the purposes of our overall study design, which was to investigate the influence of past fire regimes on abundance and diversity of biota, we concentrated on those intervals with consecutive short intervals (with '11' in the sequence) or consecutive long intervals ('33' in the sequence). For our study area, this resulted in five independent areas of successive short fire intervals, and two adjacent although unique areas of successive long intervals (Fig. 3b). Additionally, there was one area that had experienced only 2 fires in the past 30 years (in 1972 and 2002; a fire interval of 30 years), and thus had a fire interval code of '3000' (Fig. 3b). The zeroes in this code represent the fact that there is only one fire interval between 1972 and 2002 in relation to the maximum number of intervals within the dataset (which is four). Thus, retaining zeroes in a sequence provides the user with additional information about the entire dataset, adding additional strength to the methodology.

To our knowledge, this is the first representation of fire interval data in a GIS system in such a way, and represents a significant step in the way we are able to investigate complex historical datasets. A similar approach was used by Cary & Morrison (1995) to investigate the influence of fire interval patterns on floristics in sclerophyllous communities in the Sydney region, though they simply mapped fire histories within their study area. The benefits that we see of this work are that very complex or large scale data sets can be simplified at a temporal scale, and these data can be overlaid with other GIS layers to investigate patterns of fire history against information such as vegetation type, topography, climate and soils. Retrospective studies have inherent difficulties, such as assumptions that the study areas are similar in all ways other than the historical 'treatment'. By using GIS data layers, it is possible to denote areas that are similar in all physical, meteorological and geographical respects. However, ground truthing of these sites is essential.

To do this work, we were fortunate to have an accurate spatial history of fire occurrence for an area of 1.44 million hectares, which we reduced to 50 000 ha for the specific purpose of studying fire history influences on biota. Constructing similar fire interval sequences would not be possible without accurate spatial and temporal information. Missing data could have potentially large ramifications, particularly where missing fire data could mean the difference between a short interval and a long interval. However, one of the strengths of this technique is its applicability to a range of community types. This technique could be used for Canadian boreal forests, African savannas, Australian forests and other fire-dependent ecosystems if a detailed fire history and relevant life history information of major indicator flora or fauna is known (Gill & McCarthy, 1998). In some ways, this represents a potential weakness too, as the classifications are set subjectively. However, if the science is good, the classifications should be sound. There is also the scope to increase or decrease the number or form of the classifications. For example, Cary & Morrison (1995) investigated "very short" (1-3 years) fire intervals. This could be incorporated into a classification system, as could any biologically relevant class. Classification of fire intervals for a given study depends on (i) the research question, and (ii) vital attribute information of the organisms in the ecosystem (Noble & Slatyer, 1980).

The technique that we have described here for our 50 000 ha study area works best because we have a common fuel age from which to work back in time. The technique would be less useful where the study area has a variety of fuel ages, as there is no common, recent fire. For some applications, this would not be a problem (e.g. in a general description of fire histories); however, biological survey work in southwestern Australia is best undertaken in areas with an even fuel age (Bell & Koch, 1980).

Statistical analysis of fire intervals provides mean fire intervals and variation in fire interval (Gill & McCarthy, 1998), which fails to identify the successive short or long fire intervals that we considered important in our studies on fire history and biodiversity. An area that

has a low mean fire interval (suggesting frequent fire) may not have a species composition any different to an area with a higher mean fire interval (suggesting infrequent fire), if the actual intervals do not have any significant biological effects. Morrison *et al.* (1995) investigated the effects of fire intervals and time-since-fire on dry sclerophyll communities near Sydney, and concluded that variation in floristic composition is not solely related to the shortest fire interval, but to combinations of inter-fire intervals through time. We argue that it is important to recognise where fire intervals are below or above certain thresholds as well as the order in which fire intervals occur. The fire interval sequences that we have developed and described in this paper aid in identifying these thresholds and temporal patterns. Our method permits rapid investigation of the sequences of fire intervals across the landscape that can be used to investigate spatial complexity of contemporary fire intervals, and can aid in research on the impacts of fire regimes on biodiversity. When this information is linked to the original fire history database, the dataset is powerful in terms of its potential to investigate real fire regimes in the contemporary era. This type of data could have applications in a variety of research projects, particularly those that investigate fire management and biodiversity issues. The threat of climate change and its influence on fire regimes and biodiversity (both singularly and in combination) (Cary, 2002; Hughes, 2003) are possible applications of this type of data. Furthermore, the technique of classifying temporal data can be used for other variables, such as fire season, fire type (planned or unplanned), or non-fire-related data.

ACKNOWLEDGEMENTS

Thanks to Dr Neil Burrows and Bruce Ward (Science Division, DEC) for access to their Fire Response Database for flora of southwestern Australia. Craig Carpenter and Femina Metcalfe (Fire Management Services, DEC) provided invaluable advice on the creation of the Fire History Database. Principally Ray Flanagan, Jeff Bennett, Dennis Marshall, Rod Simmonds and Wolf Tiedemann from Frankland and Donnelly DEC Districts provided valuable information when TH validated the fire history information.

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PARKINFO: A GEOGRAPHIC INFORMATION SYSTEM FOR LAND MANAGERS

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Kington W.J. 2008 06 25: Parkinfo: A Geographic Information System For Land Managers. *Proceedings of the Royal Society of Queensland*, 115: 113-118. Brisbane. ISSN 0080-469X.

Queensland Parks and Wildlife Service (QPWS) an entity of the Environment Protection Agency, administers a large reserve system for which fire management is crucial. A customised agency geographic information system (GIS) known as ParkInfo, has become increasingly important to support innovations in fire policy and practice. Integrating fire zoning, fire history, vegetation and other data, ParkInfo provides a simple but powerful tool supporting fire planning, implementation and reporting.

Aimed at field-based ranger staff, many of whom have a limited knowledge of mainstream GIS, ParkInfo allows easy access to relevant GIS data through a simple to use interface. Custom built modules guide users through the process of recording fire and pest data, easing and standardising data capture. Reports are generated about individual fire or pest management activities, or about overall fire and pest management performance. Staff can visually interrogate numerous layers and undertake simple analyses aimed at aiding management. Specific modules enable the easy production of hardcopy maps and integration with global positioning (GPS) technology. The fire and pest data are extracted twice a year and merged to create a statewide database enabling reporting and analysis of trends. Significantly, ParkInfo captures fire history to aid future planning decisions.

ParkInfo has instigated organisational change within QPWS by providing access to GIS technology and providing tools for improved fire management. The application has been embraced to varying degrees by staff across the state and in many areas it is now an integral component of land management.

❐QPWS, GIS, ArcView, fire history

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ParkInfo is a GIS designed for the QPWS land manager. It is a relatively simple to use tool to assist in planning land management activities and provides a means to access, integrate and visualise spatial and related data. Modules simplify and standardise the capture of fire and pest data and reporting modules help QPWS evaluate natural resource management performance (Kington & Drake, 2005).

WHAT ARE PARKINFO'S OBJECTIVES?

The objectives of ParkInfo are summarised below (from Brown *et al.*, 2004):

- To provide QPWS land managers with a GIS that is accessible to basic computer users;
- To present a broad range of spatial data to assist in land management planning;
- To enhance park management capability by providing a powerful tool to integrate data;
- To provide a GIS with broad application for land management including infrastructure management, natural and cultural data inventories, planning and implementing land management and monitoring;
- To simplify and standardise the capture of natural resource management (NRM) data, initially focusing on fire and pests;
- To provide information about land management activities to agency information systems, to aid in evaluating park management performance; and
- Inclusion of select tools to interrogate data to aid in NRM planning.

WHY PARKINFO?

QPWS is the largest land manager in the state of Queensland, managing a vast area of 14,829,440 ha. Effective management of protected areas, including infrastructure, fire, pests, ecosystems, species and resource inventories, requires spatial information (Kington & Felderhof, 2001).

For many years access to spatial information within QPWS was variable. The transition of spatial

information to the digital realm generated inequitable access, with regional and city centres able to benefit from the technological shift, while access to contemporary spatial information by those with direct land management responsibility (park and district bases) was generally poor.

Park and district bases that began to use GIS and GPS technology did so without agreed standards and procedures, raising concerns regarding data storage, consistency, duplication of effort, communication of information and an inability to roll-up data to evaluate park management performance (Beetson, 2004).

Between 1998 and 2001 there was growing recognition among QPWS staff and executives of the need for improved business and NRM systems, and funds became available for their development. The adoption of a statewide fire policy and fire management system highlighted the need for supporting GIS. The then Queensland Department of Natural Resources had demonstrated a relevant application of technology with the PestInfo GIS system (Beetson, 2004). These pressures culminated in the development of the ParkInfo Project Plan in March 2001 (Kington & Felderhof, 2001).

HOW DOES PARKINFO WORK?

ParkInfo was developed to be operational at remote locations. Given the size of Queensland, the geographic spread and remoteness of many staff, costs and size of the data sets and inconsistent network connectivity, it has not been plausible to have a networked or web based GIS accessible by land management staff. Consequently, ParkInfo runs on individual computers or across local servers (Beetson, 2004).

ParkInfo is a customised extension of the commercial ESRI ArcView 3.3 GIS product. Aimed at field-based ranger staff, many with limited computer skills, ParkInfo substantially simplifies ArcView alleviating the need for specialist GIS skills. Modules for fire and pest management step users through an otherwise complex process for spatial and attribute data capture, allowing statewide consistency, the capture of mandatory information and recording of meta-data. The captured fire and pest data are extracted twice a year and merged into district, regional and statewide databases, enabling (mostly historic) reporting of management performance. Additional modules simplify production of hardcopy maps, labelling of features, building search queries, integrating GPS equipment and 'hotlinking' (linking non-spatial data such as reports and photographs).

Critical to the success of ParkInfo is a network of 'Configurers' supported by 'Administrators'. Configurers are intermediate GIS users responsible for developing ParkInfo 'instances', rolling the instance out to park bases, maintaining and updating instances, training field-based staff, verifying data about pests and fire, and rolling-up data (collecting and conglomerating data captured at park bases) biannually. ParkInfo instances are ready to use GIS projects covering a logical geographic area of QPWS managed land or water. Instances can include a single reserve, several reserves that are managed together as one geographical entity, or even a section of a reserve (for example, a large reserves that has its day to day management divided between disperate management bases). GIS professionals in central office and regional bases administer ParkInfo providing technical support to Configurers.

HOW IS PARKINFO USED TO SUPPORT FIRE MANAGEMENT?

A number of tools are available to aid planning, reporting and implementation of fire management. As required by the QPWS Fire Management System, users can develop and keep track of planned burn proposals and record the history of wildfires and planned burns. Users can also plan future fire management, create reports about individual fires or overall fire management performance and compile fire action maps to aid field implementation.

PLANNED BURN PROPOSALS

A custom-built module steps users through a process of adding or editing data for a burn proposal (Fig. 1). Spatial information is entered via GPS, by digitising on screen or by importing existing spatial data. Attributes are collected about burn objectives, responsible persons, estimated costs, required weather conditions, expected fire behaviour and potential risks and hazards.

PLANNED BURN PROGRAM

A planned burn program is a series of burn proposals aimed at achieving fire management goals as expressed in strategic fire plans. Land managers keep track of planned burn programs by displaying the approval status of proposals as shown in Figure 2. An annual meeting of key staff assesses the approval of burn programs.

To aid decisions with regard to priorities, timing and

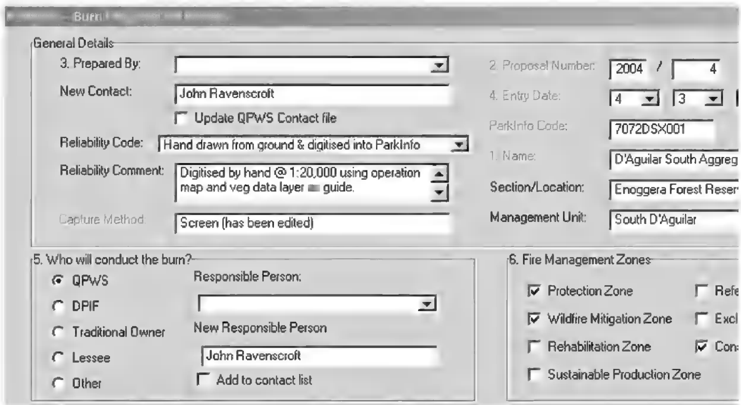

FIG. 1: Extract from the first page of the parkinfo burn proposal form. Source: South D'Aguilar ParkInfo Instance, Burn Proposal 2004/4 by John Ravenscroft

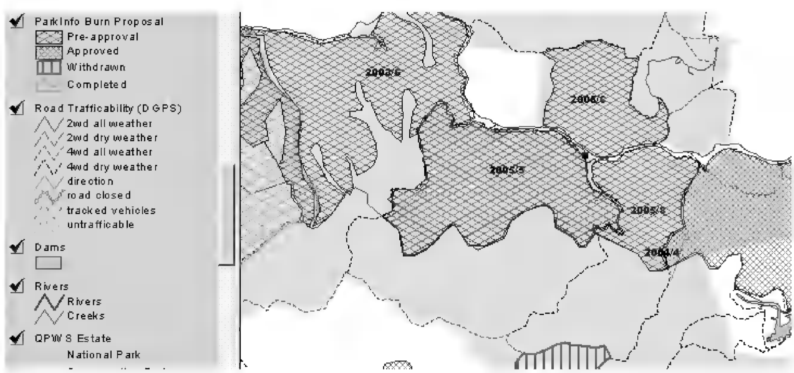

FIG. 2: Example status of burn proposals. Source: South D'Aguilar ParkInfo Instance

resources; and as an overview of upcoming planned burns, ParkInfo can generate printable summaries as shown in Figure 3.

FIRE REPORTS

Users are guided through a module for adding or editing reports about wildfires and planned burns. Information is captured about the spatial location of the fire, accuracy of the data, persons conducting fire management, costs, labour, intensity of the fire, ecosystems burnt and the results of management objectives. Reports build into a fire history for each QPWS managed area (see example in Figure 4), and are rolled-up into a state-wide QPWS fire history.

FIRE MANAGEMENT PLANNING

Fire zones within ParkInfo spatially convey the management recommendations of fire plans. When tools to interrogate fire history are compared to zoning plans, ParkInfo can assist in planning future fire management. In Figure 5, the last 6 years of fire history (in black solid fill) is contrasted against wildfire mitigation zones (diagonal hatching). Wildfire mitigation zones are intended to mitigate the spread of wildfire, and in the example provided, areas of wildfire mitigation zone that are not overlaid by recent fires (in black solid fill) are potentially overdue for planned burns.

In Figure 6, fire history (shown in solid fill) is compared to the recommended ecological fire regimes for vegetation types (shown in hatched patterns). Biodiversity data (ecologically important ecosystems

Burn Programme Summary for D'Aguilar South (7072DSX001)

as at 25 March 2006

al

eason	Name	Section/Location	Who will conduct the burn	Fire Management Zones	Proposal Area	Purposes of the burn	E S (I
utumn	D'Aguilar South Aggregation	East of 309 Barracks	QPWS	Conservation Zone	238	1 Conservation 2 Weed Management	17
utumn	D'Aguilar South Aggregation	Dunnings Break D'Aguilar Forest Reserve	QPWS	Wildfire Mitigation Zone	332	1 Hazard Reduction 2 Weed Management 3 Conservation	16
utumn	D'Aguilar South Aggregation	Between Hell hole & Centre Road	QPWS	Conservation Zone	216	1 Conservation 2 Weed Management	16
utumn	D'Aguilar South Aggregation	Between Hell hole & Centre Road	QPWS	Conservation Zone	216	1 Conservation 2 Weed Management	16
utumn	D'Aguilar South Aggregation	North of 309 Barracks	QPWS	Conservation Zone	104	1 Conservation 2 Hazard Reduction	9
utumn / Vinter	D'Aguilar South Aggregation	Between Scrub road and Black Soil Rd	QPWS	Conservation Zone	172	1 Weed Management 2 Conservation	9
ummer	D'Aguilar South Aggregation	Hollmans and Centre road	QPWS	Conservation Zone	86	1 Conservation 2 Hazard	9

FIG. 3: Extract of a planned burn program summary. Source: South D'Aguilar

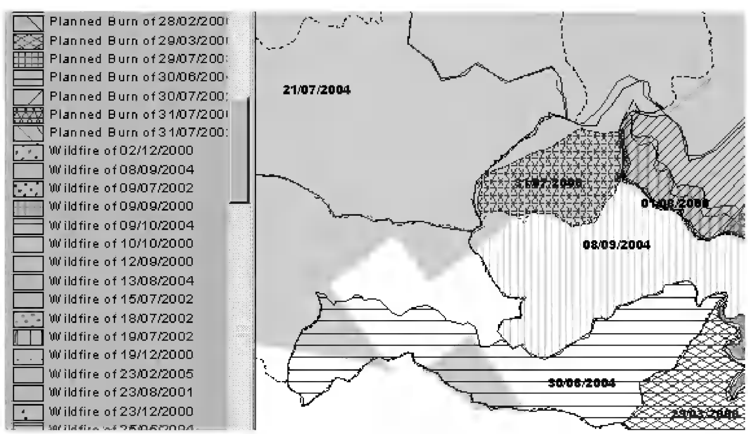

FIG. 4: Example fire history. Source: South D'Aguilar ParkInfo Instance

FIG. 5: Example application of fire history to wildfire mitigation zones. Source: South D'Aguilar ParkInfo Instance

shown in grey with no border and endangered, vulnerable and rare species by asterisks) adds guidance to burn priorities and builds an appreciation of ecological requirements and constraints. Detailed data such as roads, management points, infrastructure, creeks and contours build an appreciation of practical concerns.

SUMMARISING FIRE MANAGEMENT

Statistics summarising fire management (Fig. 7) can be generated at a park, district, regional or statewide level, providing an overview of fire management activities.

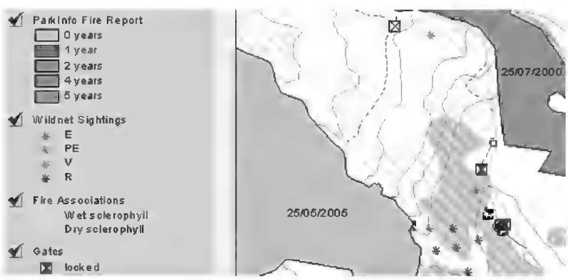

FIG. 6: Example application of fire history to conservation zones. Source: Tamborine ParkInfo Instance

FIRE ACTION MAPS

A customised module simplifies the production of printable maps, including fire action maps that are compiled to support emergency response and field operations. Fire action maps include such things as topography, property boundaries, water access points, gates, buildings, roads and vegetation grouped by flammability.

TO WHAT EXTENT HAS PARKINFO BEEN UTILISED?

There are currently 422 ParkInfo instances across the state, nearly all of which (99%) have been rolled out to local management staff. As at December 2005, 68% of these (see Fig. 8) have had fire and/or pest data entered, indicating a moderate uptake by park-based staff. As the sole indicator for usage and uptake, this figure is somewhat misleading as a number of reserves lack important fire and pest management issues so do not attract data entry effort. Looked at by area, 74% of QPWS managed estate has had fire or pest data added (see Fig. 8), indicating that larger reserves are attracting

1. Number of wildfires and planned burns:

	Wildfires	Planned Burns	TOTAL
Central	68	89	157
Northern	196	276	472
Southern	580	850	1430
Unknown	0	6	6
TOTAL	844	1221	2065

2. Total area burnt by wildfires and planned burns (ha):

	Wildfires	Planned Burns	TOTAL
Central	421455	70247	491702
Northern	6476882	1102214	7579096
Southern	615199	661179	1276379
Unknown	0	1706	1706
TOTAL	7513536	1835346	9348883

3. Total costs of fires:

	Wildfires	Planned Burns	TOTAL
Central	271156	192515	463671
[illegible]	[illegible]	[illegible]	[illegible]

FIG. 7: Extract of statistics for fire reports. Source: Patricia Hanslow, Statewide ParkInfo Instance

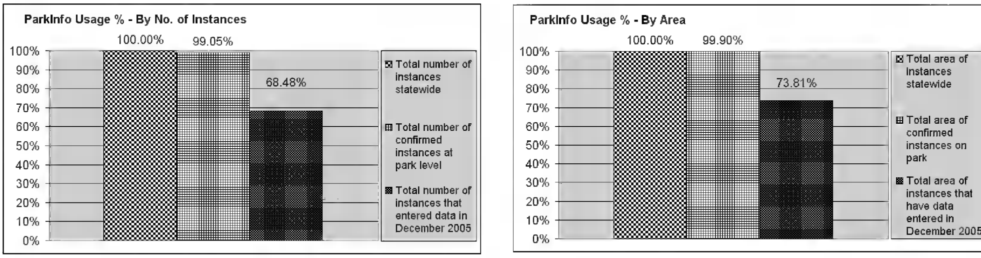

FIG. 8: PARKINFO UPTAKE STATEWIDE
Source: Patricia Hanslow, 6-3-06 based on December 2005 data roll-up

a greater effort.

Anecdotally, ParkInfo has been well received by QPWS land managers who can readily appreciate its usefulness, and many locations use it regularly to support land management. However, there are still significant gaps where shortfalls in resources, support and relevant data curtail use.

CONCLUSION

With the technological transition to digital spatial data, ParkInfo provides an accessible portal for land managers. ParkInfo has instigated organisational change by providing access to GIS technology, standardising NRM data capture, providing useful management tools and reporting functions. The application has been embraced to varying degrees by staff across the state and in many areas is now integral.

Limitations include the use of ESRI ArcView 3.3 which will not be supported by future computer operating systems, a time consuming process of data collection and updating and QPWS specific forms for data capture that limit ParkInfo's potential usefulness to other government bodies that might have different data capture requirements.

Future development provides opportunities to explore partial use of web-based platforms, modules to capture a much broader range of data not limited to fire and pests, and to provide enhanced tools to aid land and water management.

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AUTHOR PROFILE

Working in nature conservation with QPWS, a major theme of W.J. Kington's career has been translating scientific or technical information into practical conservation management systems, plans and guidelines for use by land managers and for the benefit of nature conservation. He has been active in the development of the QPWS fire system, ParkInfo, the QPWS pest system and the South East Queensland Fire and Biodiversity Consortium's monitoring guidelines and training. His particular areas of interest are fire systems, fire ecology and monitoring.

FUTURE OPTIONS FOR FIRE BEHAVIOUR MODELLING AND FIRE DANGER RATING IN NEW ZEALAND

STUART A.J. ANDERSON

Anderson S.A.J 2008 06 25: Future Options For Fire Behaviour Modelling And Fire Danger Rating In New Zealand. *Proceedings of the Royal Society of Queensland*, 115: 119-127. Brisbane. ISSN 0080-469X.

Bushfire research in New Zealand is focussed on developing a national fire danger rating system and fire behaviour prediction models. The approach has been to adapt the Canadian Forest Fire Danger Rating System to the New Zealand fire environment through empirical data collection from experimental fires and wildfires. Research has contributed to improved fire management and increased community and firefighter safety, but there are still significant gaps in the knowledge of fire behaviour in New Zealand fuels. Current research is focussed on developing fire behaviour models for fuel types not included in the Canadian system. Development of shrub fire behaviour models is a priority, given the significant proportion of bushfires in these fuels. However, this has highlighted the need to re-examine some of the fundamental principles guiding the New Zealand approach to fire behaviour modelling and fire danger rating. In New Zealand, fire behaviour prediction and fire danger rating are closely linked, compared to other countries where the two systems are separated. This can create difficulties in distinguishing appropriate spatial and temporal fire danger levels versus site-specific fire behaviour predictions. Other issues include selecting equation parameters and application of empirical systems in fuels different from those where observations were made. This paper reviews these issues and presents alternatives and options for the future.

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In New Zealand (NZ), fire danger is assessed using the New Zealand Fire Danger Rating System. The system is based upon the Canadian Forest Fire Danger Rating System (Stocks *et al.*, 1989), which was implemented in NZ in 1980 when the Fire Weather Index System module was adopted (Valentine, 1978). Coordinated development of the Canadian Forest Fire Danger Rating System at a national level commenced in Canada in 1965 (Taylor & Alexander, 2006). The system is empirically-derived (based on field data), but physical theory was applied in the development of algorithms and to supplement "empirical gaps" (Van Wagner, 1998).

The NZ Fire Danger Rating System provides reasonable assessments of fire danger in forest and grassland fuels in NZ (e.g., Anderson, 2004; Rasmussen & Fogarty, 1997; Pearce & Alexander, 1994), but difficulties have arisen in applying the system to shrub fuels. A major focus of bushfire research in NZ is the development of a fire danger rating system and fire spread models for shrub fuels, as fires in these fuels represent a considerable proportion of the annual number of fires and area burned. These difficulties have prompted the need to reconsider the philosophy behind fire danger rating and fire behaviour modelling approaches in NZ. This paper provides an overview of the NZ Fire Danger Rating System and then explores options for the future.

THE NEW ZEALAND FIRE DANGER RATING SYSTEM (NZFDRS)

The NZFDRS forms the core of a range of fire management applications, from prevention and preparedness planning through to fire suppression, investigation and operational reviews (Anderson, 2004). The NZFDRS comprises four major subsystems (Fig. 1a): the Fire Weather Index System; Fire Behaviour Prediction System; Accessory Fuel Moisture System; and the Fire Occurrence Prediction System.

THE FIRE WEATHER INDEX (FWI) SYSTEM

The Fire Weather Index System (Fig. 1b) is the core of the NZFDRS, consisting of 6 relative numerical ratings. The system is based on a reference fuel type, Canadian mature jack (*Pinus banksiana*) and lodgepole (*P.contorta*) pine stands on level terrain, not too dissimilar to mature pine plantations in NZ. Basic inputs are four weather observations (temperature,

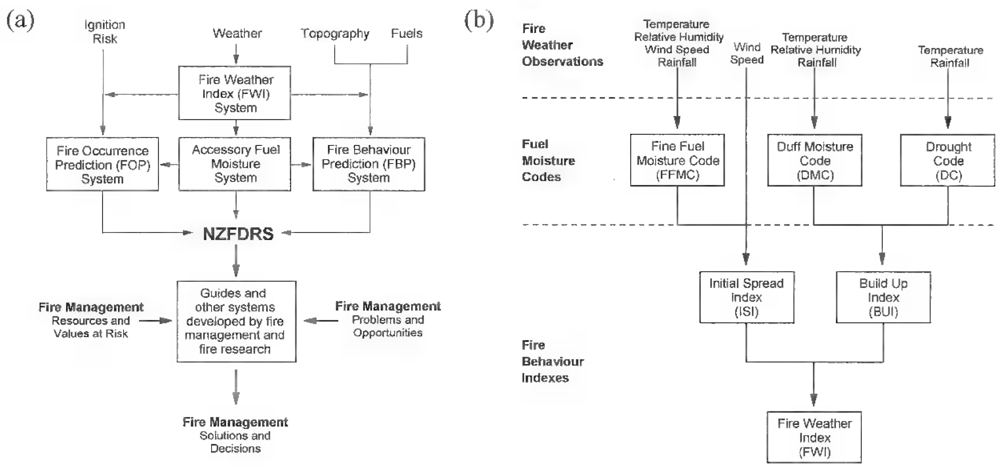

FIG. 1. Simplified structure diagrams for (a) the New Zealand Fire Danger Rating System (Fogarty *et al.*, 1998), and (b) the Fire Weather Index (FWI) System module (Anon., 1993).

relative humidity, wind speed measured at a height of 10 m above the ground, and 24-hour accumulated rainfall) measured at noon (standard time) daily. The numerical ratings are intended to represent fire danger conditions during the peak fire danger period, generally around 4pm.

These 6 numerical ratings consist of 3 fuel moisture codes and 3 fire behaviour indices. The 3 fuel moisture codes are the Fine Fuel Moisture Code, Duff Moisture Code and Drought Code. The Fine Fuel Moisture Code represents the moisture content of fine surface litter and indicates ignition potential; the Duff Moisture Code the moisture content of loosely compacted duff of moderate depth and indicates potential for smouldering and ignition from lightning strikes; and the Drought Code the moisture content of deep compact organic matter and the potential for deep-seated burning. For each of the codes, moisture is added after rain and subtracted after each day's drying. They all have built-in time lags and rainfall thresholds (below which precipitation will not lower the value). Higher values of the moisture codes correspond to lower moisture contents (Stocks *et al.*, 1989).

The 3 fuel moisture codes (and wind speed) are linked in pairs to form 2 intermediate and 1 final index of fire behaviour. The Initial Spread Index combines the effect of wind speed and fine fuel moisture content to indicate fire spread rate; the Buildup Index combines the Duff Moisture Code and Drought Code and represents the total amount of fuel available for combustion; and the Fire Weather Index combines the Initial Spread Index and Buildup Index to indicate the intensity of a spreading fire (on level terrain). A detailed description of the development and structure of the FWI System is contained in Van Wagner (1987). The system has had some modification for use in New Zealand, including altering the daylength factors and seasons in the drying codes, and providing an adjustment for latitude (Anon, 1993).

THE FIRE BEHAVIOUR PREDICTION (FBP) SYSTEM

The FBP System (Fig. 1a) describes fire behaviour characteristics and takes account of variations in fuel type and topography that are not accounted for in the FWI System. Primary outputs are the rate of fire spread, fuel consumption, head fire intensity, and fire description (e.g., crown or surface fire). Secondary outputs are head, flank and back fire spread distance, flank and back fire rates of spread and intensities, fire area and perimeter, rate of perimeter growth, and length-to-breadth ratio. These outputs are determined by prevailing weather conditions, based on wind speed and FWI System components, and on fuel type and slope. A full description of the development of the Canadian FBP System is contained in Forestry Canada Fire Danger Group (1992) and Hirsch (1993).

The development of the FBP System in NZ has followed the same empirical approach as Canada,

i.e., correlation of FWI System outputs with observed fire behaviour from experimental fires and wildfires in different fuel types. The Canadian FBP System is based on 16 benchmark fuel types, whilst in NZ 7 benchmark fuel types are currently recognised. Limited validation of fire spread models for the mature pine plantation, logging slash and pasture grassland fuel types has been undertaken, but many of these models require further validation (Pearce & Anderson, 2008).

FIRE OCCURRENCE PREDICTION SYSTEM

The Fire Occurrence Prediction System predicts fire occurrence from both human and natural causes, and is under development in Canada (Taylor & Alexander, 2006). No work has commenced on its development, and it is not in use in NZ. A particular problem to date in NZ has been a lack of adequate data containing both wildfire cause and fire environment information.

ACCESSORY FUEL MOISTURE SYSTEM

The Accessory Fuel Moisture System supports special requirements of the FWI, FBP and Fire Occurrence Prediction Systems, such as fuel-specific moisture codes, adjustments for aspect and diurnal trends in FWI System values (Stocks *et al.*, 1989). It is incomplete in Canada with no development in NZ, although Canadian diurnal FWI calculation routines have been adopted in NZ (Taylor & Alexander, 2006; Lawson *et al.*, 1996; Alexander *et al.*, 1984).

FIRE DANGER CLASSES

Three benchmark fuel types are used for fire danger rating in NZ (Forest, Grassland and Scrubland). The Forest and Grassland Fire Danger Class Criteria are based on the Canadian fuel types C-6 (Conifer Plantation) and O-1b (Standing Grass) respectively (Alexander, 1994). The Scrubland Fire Danger Class Criteria have been subsequently developed in NZ (NZ Fire Research, 2000b). The 5 fire danger classes (Low, Moderate, High, Very High and Extreme) are based upon fire intensity and indicate suppression difficulty (Alexander, 1994).

FIRE DANGER RATING SYSTEMS

The three major fire danger rating systems in use around the world are the United States (US), Canadian and Australian systems. Fire danger rating systems are designed to provide a broad-area rating of fire danger in terms of expected burning conditions. These outputs are used for public information and area-based fire management. Fire behaviour prediction systems provide indications of fire behaviour at the site-specific level, for actual fires on the landscape (Chandler *et al.*, 1983). The Canadian and NZ fire danger rating systems are conceptually different to the US National Fire Danger Rating System (Deeming *et al.*, 1974; Deeming *et al.*, 1977). The US National Fire Danger Rating System completely separates the fire danger rating and fire behaviour prediction systems, and they are determined through different inputs and computations (Andrews, 1988). The Canadian system does distinguish between fire danger rating and fire behaviour prediction, but not as clearly as the US system. The Canadian system uses the FWI System indices for broad-area fire danger rating in benchmark fuel types, and the FBP System caters for changes in fuel type and topography for site-specific fire behaviour prediction. However, the inputs for the FBP System rate of spread models are based on correlations between observed rate of fire spread and FWI System indices, so that the two systems are not as distinctly separate as in the US system. With an inadequate understanding of the different purposes of the two systems, confusion can arise when trying to separate broad-area fire management from site-specific fire behaviour prediction. The two major forest fire danger rating systems in Australia, the McArthur Forest Fire Danger Rating System and the Western Australia Forest Fire Behaviour Tables, also do not clearly separate fire danger and fire behaviour (Luke & McArthur, 1978; Burrows & Sneeuwjagt, 1988; Cheney, 1988; Sneeuwjagt & Peet, 1998; San-Miguel-Ayanz *et al.*, 2003). However, the results of the Project Vesta fire behaviour experiments in dry Eucalypt forest will lead to improvements in forest fire behaviour models and recommendations to separate elements of fire danger rating and fire behaviour prediction (Gould *et al.*, 2007). The revised grass fire spread models in Australia have led to a separation of fire behaviour and fire danger rating (Cheney & Gould, 1995; Cheney & Sullivan, 1997; Cheney *et al.*, 1998).

SHRUBLAND FIRE BEHAVIOUR RESEARCH IN NEW ZEALAND

Shrub vegetation is one of three dominant plant formations in NZ, along with grassland and forest. Pure shrub communities cover approximately 4% (1.1 million ha) of the land area of NZ, with a further 24% (6.5 million ha) consisting of shrub-grassland or shrub-forest mixtures (Newsome, 1987). Fires in shrub fuels account for approximately 30% of all vegetation fires and 43% of the total annual area burned (National Rural Fire Authority, Annual Return of Fires). Fire

has also been associated with the development of shrublands, especially since human settlement began approximately 1000 years ago (Burrows *et al.*, 1979).

Shrubland species and communities around the world are renowned for their flammability and ability to burn at very high rates of spread and extreme fire intensities, under levels of fire danger that would not be considered extreme for other fuel types (Fogarty, 1996; Fernandes, 1998; De Luis *et al.*, 2004). This is probably due to factors such as the presence of volatiles and other chemical compounds in foliage, and a typically high elevated dead fine fuel loading and low fuel bulk density (well-aerated and exposed to solar radiation). Many shrubland species are also dependant on fire for regeneration (Specht, 1979; Catchpole *et al.*, 1998). Alexander (1994) describes fires in shrubland as being determined by an "on/off" switch in terms of conditions where no ignition and fire spread will occur, compared to rapid fire development with extreme rates of spread and intensity. There does not appear to be any intermediate or slow development of fire behaviour in NZ shrub fuels.

SHRUBLAND FIRE BEHAVIOUR

Attempts have been made to develop a rate of fire spread (ROS) model for NZ shrub fuels, following the Canadian approach. This is based on an S-shaped asymptotic curve to model rate of spread, using the general equation shown below (Equation 1):

$$\mathrm{ROS} = a \times \left[1 - e^{(-b \times ISI)}\right]^c \qquad (1)$$

where a, b and c are parameters specific to a fuel type, and ISI is the Initial Spread Index value of the FWI System (Forestry Canada Fire Danger Group, 1992). This general equation was visually fitted to NZ data from 29 experimental burns and 3 wildfires, with values of a, b and c derived of 4920, 0.1 and 1.5 respectively (Fig. 2). It is apparent that the model fits the data rather poorly, with an R^2-value of 0.25. In fact, a basic linear relationship describes the relationship between Initial Spread Index and rate of spread more satisfactorily (R^2-value of 0.51), although it does not have the sigmoidal shape which represents the rapid initial increase in rate of spread and subsequent levelling-off at very high values of Initial Spread Index (Forestry Canada Fire Danger Group, 1992). The lack of data in the Initial Spread Index range of 10-20 is immediately apparent, with only 2 data points having values of greater than 8 (NZ Fire Research, 2000b). It would be reasonable to question any relationship if these two data points were removed, and it is also quite clear that considerably more data points at these higher values of Initial Spread Index are required.

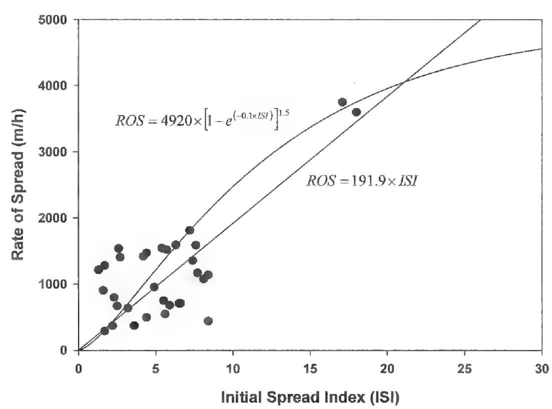

FIG. 2. The New Zealand rate of fire spread (ROS) model for shrub fuels (sigmoidal form), compared against a basic linear relationship fitted to the data.

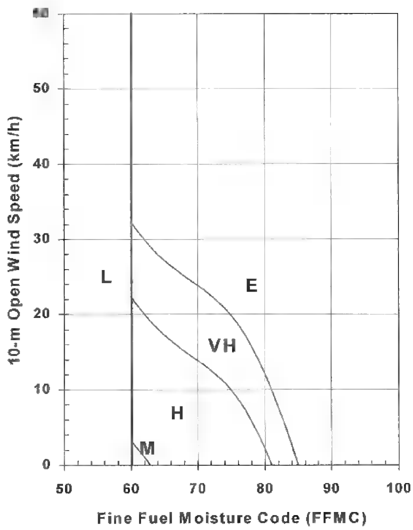

FIG. 3. The NZ Scrubland Fire Danger Class.

SHRUBLAND FIRE DANGER RATING

The first version of the NZ Scrubland Fire Danger Class Criteria was released in 2000 (NZ Fire Research, 2000a) and subsequently revised to its current format (NZ Fire Research, 2000b). The system uses the Fine Fuel Moisture Code of the FWI System and the 10-m wind speed to determine fire danger classes (Fig. 3). This is effectively the Initial Spread Index (Fine Fuel Moisture Code combined with wind speed, refer Fig. 1). The system also assumes a standard shrub fuel load of 20 t/ha and includes a Fine Fuel Moisture Code threshold of 60, below which the fire danger class is always Low. The standardised fuel load of 20 t/ha is based on fuel models developed for NZ shrub fuels and predicts available fine fuel (dead fuels less than 5 mm and live fuels less than 2 mm in diameter) (Fogarty & Pearce, 2000). This assumes a standard fuel load across the country for fire intensity calculations (upon which the fire danger classes are based), and removes the need for fire managers to assess fuel loads across the country for different areas of shrub fuels. The threshold Fine Fuel Moisture Code value of 60 provides an improved response to moist conditions over the previous version where Very High or Extreme fire danger was possible at high fuel moisture levels. However, this system presents many problems. Extreme fire danger is still predicted too frequently, e.g., a recent study found that the average annual number of days of Very High and Extreme fire danger in shrub fuels across the country ranged from 104 – 294 days (Pearce *et al.*, 2003). The assumption of a standard fuel load of 20 t/ha across the country and the Fine Fuel Moisture Code threshold of 60 require further validation (Pearce, 2001).

DISCUSSION AND CONCLUSIONS

The NZ Fire Danger Rating System works reasonably well for forest and grassland fuel types. Adopting and modifying the Canadian Forest Fire Danger Rating System to NZ has been more effective than developing a new system (Fogarty *et al.*, 1998). However, recent efforts to develop a fire spread model and fire danger rating system for shrub fuels have highlighted some shortcomings in this approach, and prompted the need to reconsider the way forward. Results from the attempts to develop a shrub rate of spread model, based on the FWI System components without any modification of the codes and indices to represent fuel moisture and fire behaviour in shrub fuels, indicate that this approach is not appropriate. This is probably partly due to the fact that fire behaviour is being modelled using FWI System indices that are based on a reference fuel type of needle litter on the forest floor under a mature conifer overstorey. NZ shrub fuels have a different structure to that of a mature conifer forest, with large amounts of dead elevated material subject to considerably faster rates of drying than conifer needle litter on the forest floor. Another likely reason is that the relationship between spread rate and fuel moisture content varies with fuel type (the NZ fire spread model combines different shrub fuels). More shrub fire behaviour data also need to be collected. Alternative approaches could include modelling rate of spread using the basic parameters of fuel moisture, wind speed and fuel characteristics, rather than composite indices as contained in the FWI System (e.g., Catchpole *et al.*, 1998). The same reasons are likely for the poor performance of the NZ Scrubland Fire Danger Rating model, and also that model assumptions, such as the threshold Fine Fuel Moisture Code value of 60 and the standard fuel load of 20 t/ha values, have not been completely validated.

It is not appropriate to apply an empirical system to conditions beyond those under which the original data were collected without further validation or modification. An empirical system does not readily lend itself to major changes in the underlying functions, although in the case of NZ it may be possible to revise the existing functions within the Fine Fuel Moisture Code to accurately reflect the drying rates of shrub

fuels (e.g., Fogarty *et al.*, 1998). The Accessory Fuel Moisture System also provides the opportunity to incorporate shrub fuel moisture dynamics.

The distinction between fire danger rating and site-specific fire behaviour prediction is not as clear in the Canadian/NZ system as it is in the US National Fire Danger Rating and Australian Grassland Fire Danger Rating Systems. If users do not properly understand the key differences in the applications of fire danger rating and fire behaviour prediction systems, this can lead to confusion in their correct application. Users must understand that fire danger rating system outputs can only be applied over broad areas for activities such as determining fire season status, imposing restrictions, determining preparedness levels and public education. Fire spread models are to be used for prediction of spread rate for fires at specific points in the landscape (Cheney & Gould, 1995). In the case of the NZ Scrubland Fire Danger Class Criteria, criticism is often levelled that the outputs cannot be applied regionally because the fire danger classes are too often Very High or Extreme. Users need to recognise that the shrub fire danger class system is best used to supplement the knowledge of fire danger classes in Forest and Grassland fuel types, given that shrub fuels often occur in patches throughout the country (NZ Fire Research, 2000a).

NZ is in some respects in a fortunate position in that shrub fire spread and fire danger modelling is in a relatively early stage. This provides the opportunity to consider different approaches, as the current models have yet to gain widespread acceptance. The simplest approach would be to continue the empirical method of the Canadian system, focussing on modifying functions in the fuel drying and fire behaviour relationships to better reflect conditions in shrub fuels. The current NZ Fire Danger Rating System works reasonably well for forest and grassland, and considerable effort has been invested in establishing systems to determine daily fire danger and train users in the application of the system. Modifying an existing system is simpler and more cost-effective than developing a completely new system, and is perhaps more appropriate for NZ. A completely new approach would require considerable research effort and education of users, and is possibly beyond the resources of NZ (Fogarty *et al.*, 1998). There is a considerable amount of research still to be undertaken related to shrub fuels and fire, with better understanding of shrub fuel moisture relationships, fire behaviour and fuel characteristics required. Perhaps the greatest opportunity lies in combining empirical and physical modelling approaches, utilising models of the physics of moisture gain and loss in fuels and fire development and spread, and combining them with and validating them against empirical data.

Understanding of fire behaviour in NZ shrub fuels is improving, with data collection continuing. Collaboration through the International Heathland Fire Behaviour Modelling Group (Catchpole, 1999) is being enhanced through a focus on shrub fire behaviour in Australia and NZ, as well as research in Europe and the USA (e.g., Fernandes *et al.*, 2000; Fernandes, 2001; Viegas *et al.*, 2002; Weise *et al.*, 2004; Weise *et al.*, 2005). A major research project underway in the Australian Bushfire Cooperative Research Centre is focussed on shrub fire behaviour (Myers *et al.*, 2005). Collaboration between NZ's Scion and Australia's CSIRO Bushfire Dynamics and Aplications Group has significantly enhanced Australasian bushfire research capability. There are currently a number of projects in NZ and Australia studying fuel moisture, ignition and fire behaviour in shrub fuels. Outcomes from these research efforts will result in improved systems of rating fire danger and predicting fire behaviour in NZ and Australian shrub fuels.

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SPATIAL PATTERNS OF FIRE BEHAVIOUR IN RELATION TO WEATHER, TERRAIN AND VEGETATION

KATE A. HAMMILL AND ROSS A. BRADSTOCK

Hammill, K.A. & Bradstock, R.A. 2008 06 25: Spatial Patterns Of Fire Behaviour In Relation To Weather, Terrain And Vegetation. *Proceedings of the Royal Society of Queensland*, 115: 129-135. Brisbane. ISSN 0080-469X.

Understanding fire behaviour in different weather conditions and across large, flammable landscapes is important for fire management. In this study, the influences of weather and major landscape variables on fire behaviour were examined following a large fire in the Blue Mountains, near Sydney. Patterns of fire behaviour were inferred from a fire severity map derived using remote sensing and field validation. Fire weather on the day of burning was determined for different parts of the landscape using bureau of meteorology data and fire spread maps compiled during the event. Relative proportions of the landscape burnt by different fire behaviour classes (particularly crown and understorey fires) were determined in a geographic information system. The influence of vegetation type, fuel age and terrain on fire behaviour during two contrasting weather conditions (extreme and moderate fire weather) was examined. The analysis showed that during extreme weather, fire behaviour was dominated by either a crown fire that consumed the canopy or a fire of an intensity that scorched the canopy leaves. In relatively moderate weather, crown fire was almost non-existent and the canopy remained intact over about half of the landscape. Fuel age (time since last fire) of between 1-4 years appeared to moderate fire behaviour relative to fuel ages of 5 to >20 years. Ridge tops and slopes of 15° or less appeared to suffer more crown fire than gullies and slopes of >15°. Surprisingly, aspect did not greatly influence fire behaviour despite strong, directional winds. An important ecological implication may be that fires that occur during severe weather lead to greater landscape homogeneity than fires that occur during more mild weather

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Variations in environmental conditions (weather, terrain, vegetation) during large, multi-day fires results in complex patterns of fire behaviour across a landscape (Simard, 1991; Catchpole, 2002). Burn patterns, produced by different fire behaviours may be detected and mapped using remote sensing (e.g. ; Bowman *et al.,* 2003; Chafer *et al.,* 2004). Such patterns are also referred to as fire severity patterns. They enable us to retrospectively examine spatial and temporal variations in fire behaviour. Using a landscape-scale view of fire behaviour generated by remote sensing, questions may be explored such as 'how does weather on the day influence the occurrence of crown fire', 'how does a fire behave in different fuel types and terrain under the same weather conditions' and 'what is the spatial arrangement of patches of different fire severity (ie. what size, how close and in what part of the landscape do patches of different fire severity occur)'. Understanding these issues may be useful for fire management by improving predictions about fire behaviour, determining the influence of fuel load and terrain on fire behaviour, and understanding the condition of the post-fire landscape in which plants and animals must survive (e.g. Bradstock *et al.,* 2005).

This paper describes a simple quantitative analysis of fire severity patterns resulting from a large fire affecting part of the Blue Mountains, near Sydney, in the summer of 2001/02. The landscape in which this fire occurred is characterised by rugged, sandstone terrain and variable vegetation. The fire burnt during contrasting weather conditions over many days. Satellite imagery obtained that summer had been used to produce fire severity maps for a number of fires in the region (Chafer *et al.,* 2004). The availability

of such data provided an opportunity to examine fire severity patterns across the landscape, and thus infer fire behaviour in relation to weather (forest fire danger index), vegetation type and fuel age (time since last fire), and terrain (slope and aspect). Here, we present part of an ongoing study which aims to determine the influence of different environmental variables on fire behaviour in the Blue Mountains. Further work using the fire severity data will involve the development of a statistical model to explain and predict fire behaviour in these complex sandstone landscapes.

MATERIALS AND METHODS

STUDY AREA

The study area is located within Blue Mountains National Park, approximately 50 km west of Sydney, south-eastern Australia. This area consists of dissected terrain formed by erosion of an ancient (Triassic) sandstone plateau (Pickett & Alder, 1997). The cliff lines and gorges that characterise the area have been eroded by water over the last 20 million years, and rivers now occupy the bottom of deep v-shaped valleys. Vegetation in the area is dominated by sclerophyll shrublands and eucalypt forests (Keith & Benson, 1988; Benson, 1992). During December 2001-January 2002, a number of large fires burnt simultaneously across the region, affecting a considerable proportion of the landscape. This study examined fire behaviour patterns of one of these fires, the Mount Hall fire, which burnt about 46 000 ha of the Park after a lightning ignition in a remote part of the Park on 24 December 2001. The fire burnt for 3 weeks through rugged terrain and varied vegetation, across an altitude range of 50 m to 900 m above sea level.

FIRE AND LANDSCAPE DATA

Fire severity data used in this study were derived from the SPOT (Satellite Pour l'Observation de la Terre) 2 satellite sensor. The method used to map fire severity patterns using this data was based on a vegetation greenness index, the normalised vegetation difference index (NDVI). Patterns of Δ NDVI between a pre-fire and a post-fire image were used to map fire severity following field validation (ie. field sampling was used to determine which Δ NDVI values represent which levels of fire severity on the ground). This method has been described in detail elsewhere (Chafer *et al.*, 2004; see also Hammill & Bradstock, 2006). The mapped fire severity patterns are shown in Figure 1.

Areas burnt during different types of fire weather were determined using a combination of fire progression maps produced by the incident management team at the time of the fire (Department of Environment and Climate Change NSW, unpublished data) and Bureau of Meteorology data. Initially, on the 25 December 2001, the main fire front spread rapidly through a forest-dominated landscape driven by hot, dry westerly winds (35°C, ~8% relative humidity, gusts to ~80 km h^{-1}). The maximum forest fire danger index (FFDI, range: 0-100, McArthur, 1967) reached 100 on this day. Moderate weather (FFDI <20) and moist, south-easterly winds occurred on subsequent days, pushing the 25 km northern flank to the north and west towards the urban areas along the highway. During a return of extreme weather (maximum FFDI of 40-90) from 1-3 January 2002, the fire reached the urban-bushland interface in the central mountains. Subsequently, the fire spread into higher altitude areas (700-900 m a.s.l.) to the west where shrubland and low woodland vegetation dominates. Following rain, the fire was extinguished on around 7 January 2002. By combining the fire spread maps with weather information, we have identified parts of the landscape burnt during different weather conditions. This paper focuses on a comparison of areas affected by two of these contrasting weather conditions: (i) the area burnt by the main headfire (FFDI ~100) and (ii) the area burnt during more moderate weather (FFDI ~20) (Fig. 1).

The vegetation in the study area varies from single-layered sedge-swamps and heath (1-4 m) to multi-layered woodlands and forests of varying height (10-50 m). These vegetation types are comprised primarily of various combinations of sedges and other monocots, sclerophyll shrubs and, in woodland and forests, a canopy dominated by eucalypts. Typically, shrubland and woodland occurs on ridges, upper slopes and headwater valleys taller forests occur on mid and lower slopes and in gullies. In this study, vegetation was grouped into three broad types on the basis of structure and height: shrubland (sedge-swamp and heath), sclerophyll woodland/open-forest (trees 10-25 m tall), and tall forest with mesic understorey (trees often >30 m tall). Spatial distributions of these broad vegetation types were derived by pooling vegetation classes identified in recent mapping of the area (Tindall *et al.*, 2004). A digital grid layer representing these distributions across the study area was used in the geographic information system (GIS) analyses.

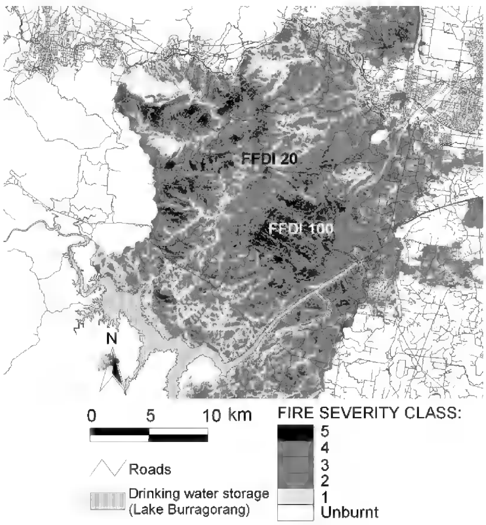

FIG. 1. Severity map of the Mount Hall fire derived using satellite imagery (see Chafer *et al.*, 2004).

Fire severity and inferred fire behaviour classes are described in Table 1. The dominant forest fire danger index (FFDI) and direction of fire spread (arrows) during two main stages of the fire progression are shown.

Number of years since the most recent fire (prior to the Mount Hall fire) was derived from historical fire maps (Department of Environment and Climate Change NSW, unpublished data) and used as a surrogate for fuel condition. A GIS layer of time since last fire, which varied in different parts of the landscape from 1 year to more than 25 years, was classified using the following categories: 1-4 years, 5-10 years, 11-20 years, >20 years.

Terrain (slope and aspect) classes were derived using a digital terrain model (DEM) and GIS data representing the location of water courses. The DEM was classified into a grid representing the following five slope classes: ridge (≤5°), 6-15°, 16-25°, >25° and gully. Typically, the more moderate inclines occur on the shoulders of ridges and upper slopes, while steeper inclines are further down in the characteristic v-shaped valleys. Gullies were defined using a GIS manipulation of the water course data, in which drainage lines were expanded to a width of 50 m and then incorporated into the classified DEM layer. The DEM was also used to derive a layer representing four aspect classes: north (315-45°), east (45-135°), south (135-225°) and west (225-315°). Ridges and gullies (as defined above) were excluded from the aspect layer.

DATA ANALYSIS

Analysis of the spatial datasets was done using Arcview GIS version 3.2 software. The landscape datasets were converted to grid format with 10 m cell size and spatially aligned with the fire severity map. Fire severity data were intersected with the data for each landscape variable using a 'combine grids' function in the Spatial Analyst toolset. The output data were used to calculate the total area and percent of landscape in each severity class in each of the landscape variable categories. The influences of vegetation type, fuel age, slope and aspect on patterns of fire behaviour during the two contrasting weather conditions are presented in graphical format here.

RESULTS AND DISCUSSION

The fire severity classes detected using remote sensing, and the inferred fire behaviour classes, are described in Table 1. During the main headfire (FFDI ~100), the canopy was consumed or scorched over the majority (91%) of the landscape. During milder weather (FFDI ~20), crown fire was almost non-existent, however canopy scorch occurred over 43% of the landscape and areas where the canopy remained unburnt comprised about 53% of the landscape. Under both weather conditions, very little of the landscape remained unburnt (1-2%) (Table 1).

During extreme fire weather (FFDI ~100), most (about 70%) of shrubland and woodland/open-forest vegetation was subject to either a crown fire or a high-intensity understorey fire and only about 5% was burnt by a low-intensity/patchy fire. In contrast, under the same extreme weather conditions, close to 40% of tall (mesic) forest was burnt only in the understorey (Fig. 2a, left). During more moderate weather (FFDI ~ 20), a relatively small proportion of shrubland and woodland/open-forest areas was burnt by crown or intense understorey fire (< 20%) and about 50% burnt at lower intensity (Fig. 2a, right).

In general, fire behaviour was more severe in shrubland and woodland/open-forest than in tall (mesic) forest, with larger proportions of the former subjected to crown fire or canopy scorch than the latter, irrespective of weather conditions. This difference may have been influenced by vegetation structure and floristics, since shrubland and woodland/open-forest are dominated

TABLE 1. Remotely-sensed fire severity classes, inferred fire behaviour classes, and percent of landscape burnt in each fire behaviour class during two contrasting weather conditions (forest fire danger index, FFDI). Patterns of these fire severity classes across the area affected by the Mount Hall fire are shown in Figure 1. Data derived from Chafer *et al.* (2004) and Hammill and Bradstock (in press).

Fire severity class (observed)	**Fire behaviour class (inferred)**	**% Landscape burnt**	
		FFDI ~100	*FFDI ~20*
5. Crown and understorey leaves consumed	crown fire	20	1
4. Crown scorched, understorey leaves consumed	intense understorey fire	51	15
3. Crown and understorey scorched	understorey fire	20	28
2. Crown intact, understorey scorched	low-intensity understorey fire	4	23
1. Crown intact, understorey partly scorched	patchy understorey fire	4	30
Unburnt	no fire	1	2

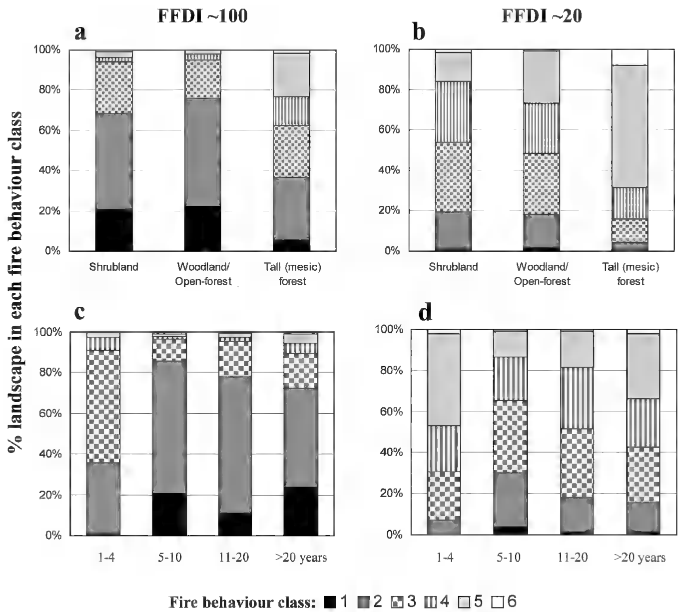

FIG. 2. Percent of the landscape affected by different fire behaviour classes within (a, b) vegetation type and (c, d) fuel age during contrasting weather conditions (a and c: FFDI~100, b and d: FFDI~20). Details of fire behaviour classes are given in Table 1. The three vegetation types (shrubland, woodland/open-forest and tall forest) represent broad structural categories, within which a number of mapped vegetation classes (see Tindall *et al.*, 2004) have been pooled. Shrubland includes sedge-swamp and heath, woodland/open-forest includes a variety of eucalypt-dominated communities with a variable sclerophyll, shrubby understorey, and tall (mesic) forest includes tall, open eucalypt forest, riparian forest and rainforest. Fuel age is the number of years since the most recent fire (prior to the Mount Hall fire) and may be used as a surrogate for fuel load.

by highly flammable species (sclerophyll shrubs), while tall forest is comprised of many mesic species, especially in the understorey where ferns, herbs and broad-leaved shrubs are common. Also, the tall (mesic) forests occur in gullies and on lower slopes and may have been more sheltered from the high winds driving the worst fire behaviour.

During extreme fire weather (FFDI ~100), young (1-4 years) fuel age was associated with the almost total absence of crown fire, however high intensity understorey fire (causing canopy scorch) still affected about one third of the young fuel areas (Fig. 2b, left). Also during extreme weather, 70-90% of the landscape with older fuels (5+ years) was affected by either crown fire or intense understorey fire (Fig. 2b, left). A similar, although less marked, trend of more moderate fire behaviour in young fuels was found during relatively mild weather (Fig. 2b, right).

In general, while the extent of crown fire during extreme weather was reduced by young fuels, fuel age had a negligible effect on complete canopy scorch (>90% of the landscape, irrespective of fuel age). The extent of unburnt and low-intensity/patchy understorey fire was hardly affected (10% of the landscape, irrespective of fuel age). Could these fuel-related effects on fire behaviour be important for fire management? For instance, is it possible to control a fire of an intensity that scorches the tree canopy, since this level of fire behaviour still dominates in young fuel during extreme weather? Other observations within the study landscape of height of leaf consumption above ground (e.g. Cheney, 1981) indicated that fire intensities beyond known thresholds for effective suppression were achieved in all fuel ages except sites with fuel ages of less than 1 year (Bradstock & Cohn, unpublished data). Also, is a fuel age of 1-4 years achievable for management if required over large areas? Maintaining such young fuel age would have major implications for biodiversity conservation, since a fire interval of 4 years is shorter than the juvenile period of many plant species in these landscapes (Keith, 1996; Bradstock & Kenny, 2003).

During extreme weather (FFDI ~100), fire behaviour was most severe on the ridges and moderate slopes: crown fire affected about 30% of ridges and <15° slopes but less than 20% of gullies and 16+° slopes. The distribution of less severe fire behaviour was similar across all slope classes, except that gullies had a greater proportion of low-intensity and patchy

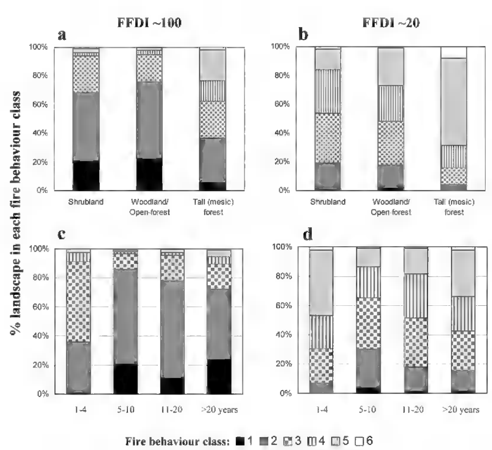

FIG. 3. Percent of the landscape affected by different fire behaviour classes on different (a, b) slopes and (c, d) aspects during contrasting weather conditions (a and c: FFDI~100, b and d: FFDI~20). Details of fire behaviour classes are given in Table 1. The distribution of slope and aspect classes across the study landscape were determined by classification of a digital elevation model (DEM) grid layer combined with spatial data on the location of water courses. Gullies are defined as a 50 m strip spanning water courses. North, east, south and west aspects are restricted to slopes >5° (ie. ridges and gullies are excluded from the aspect data).

understorey fire (about 20%) than slopes and ridges (5-10%) (Fig. 2c, left). During more moderate weather (FFDI ~20), low-intensity understorey fire was more noticeably affected by slope, with 10-20% of ridges and moderate slopes (<15°) and about 30-50% of steeper slopes and gullies being affected (Fig. 2c, right). The converse of this is that intense fire occurred over far less of the steep (16+°) slopes and gullies than on the ridges and <15° slopes.

Aspect appeared to have little influence over the distribution of the different fire behaviour classes in both extreme and moderate weather. There were roughly similar proportions of each fire behaviour class on north, east, south and west facing slopes (Fig. 2d).

These results need to be considered in view of the inter-dependence of vegetation/fuel and terrain. For instance, steep slopes in the Blue Mountains are characterised by rock outcrops and discontinuous vegetation, vegetation of shorter stature often occurs on the ridges and upper slopes (as compared with taller forest on lower slope and in gullies), and south and east aspects are often characterised by more mesic species (even on upper slopes) due to less exposure to the north-oriented sun. Apparent effects of terrain on fire behaviour are therefore likely to be influenced by these terrain-associated vegetation patterns and fuel characteristics.

SUMMARY

This study found that, during extreme (FFDI ~100) head fire conditions, the majority of vegetation was either consumed or completely scorched. There were only small areas where the fire had remained at shrub level, and almost no areas were left unburnt. Areas of unburnt canopy were restricted to tall, mesic forests and steep slopes and gullies under these conditions. Crown fire behaviour was moderated to some extent by young fuel age, however the total area with complete canopy scorch was not reduced. In contrast, during more moderate weather (FFDI ~20), crown fire was almost non-existent and the canopy remained intact over about half of the landscape. Surprisingly, aspect appeared to have little influence on fire behaviour under both weather conditions. These patterns are of relevance to fire management in terms of predicting fire behaviour and understanding its effects in fire-prone landscapes such as the Blue Mountains. Similar analyses of other large fires would be useful to further explore these findings.

ACKNOWLEDGEMENTS

We appreciate the use of remote sensing data supplied by Chris Chafer and colleagues at the Sydney Catchment Authority. Staff of DEC (NSW) Parks and Wildlife Division at the Blue Mountains Region assisted by providing GIS data of fire progression and incident reports of the Mount Hall fire. This research was funded by the Special Areas Plan of Management Research Program RD08 during 2003-2004.

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A NEW FUEL LOAD MODEL FOR EUCALYPT FORESTS IN SOUTHEAST QUEENSLAND

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Gilroy, J. & Tran, C. 2008 06 25: A New Fuel Load Model For Eucalypt Forests In Southeast Queensland. *Proceedings of the Royal Society of Queensland*, 115: 137-143. Brisbane. ISSN 0080-469X.

The behaviour of a fire, whether it is a wildfire or a prescribed burn, will largely be affected by the quantity of fine fuels present within the system. It is vital to keep track of the development of fuel in order to manage the risk and provide useful indicators for planned hazard reduction programs as well as assisting in determining ecological thresholds. Two widespread regional vegetation ecosystems (*Eucalyptus racemosa* open woodland and *Corymbia citriodora/Eucalyptus major* open forest) were studied within Redland Shire of coastal southeast Queensland. At 21 sites, surface fine fuel load were collected and measured against time since fire, rainfall (one-year post-fire), foliage projective cover and fuel depth. It was found that there was a linear relationship between time since fire, foliage projective cover, fuel depth and total fuel quantity. This was found to be a reliable model, explaining 68% of the variation in the data. For land managers, we have provided a useful model (Fuel Load = [0.286]Fuel Depth + [0.321]Time Since Fire + [0.100]Foliage Projective Cover) that can be used to quickly (as no transformation is required) and accurately determine surface fine fuel. However, we acknowledge that a linear relationship cannot be maintained between fuel quantity and the measured variable and further study is warranted to determine the thresholds of this model. Further work on fuels and fire management in southeast Queensland is also recommended.

❒ Fuel load, ecological fire, fuel management, southeast Queensland.

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Fuel load accumulation is a result of many separate and related influencing factors and as such are complex systems that are difficult to completely explain (Gill, 1997). The total fuel load at any one site will depend on the rates of accession and decomposition of litter, the vegetation type, productivity of the overstorey and understorey, the density of the vegetation, environmental conditions of the area, the fire history and fire regime (Conroy, 1993; Chatto, 1996; Millar & Urban, 2000). The accumulation of fuel, and models based on fuel quantity, rates of decomposition and time since the last fire are considered to be an essential part of the decision making process for land management agencies when estimating and predicting fuel load quantities (Fernandes & Botelho, 2003; Gould, 2006).

The most commonly used model for fuel accumulation is Olson's (1963) negative exponential model to quantify litter accumulation (Sandercoe, 1990). Two predictive models developed by McCarthy & Tolhurst (2001) were based on the Fire Danger Index (FDI) and overall fuel hazard under Australian conditions. Both models predict that as fire danger increases the benefits of a previous fuel reduction burn starts to reduce (Tolhurst *et al.,* 1992). McCarthy & Tolhurst (2001) predicted that a hazard reduction burn reducing potential fuel will only play a role in helping to reduce the severity of the fire and in assisting fire fighters with suppression for the first 4 years postburn. This research has led to fire management strategies involving the manipulation of fuel and frequency of fire. Even so, as previously mentioned, these models are based on site specific conditions, which are as yet undetermined for southeast Queensland (SEQ).

One of the questions that many land managers of SEQ ask when using fuel estimation models and guides derived from other locations of Australia is - are these models applicable to subtropical SEQ conditions? The SEQ climate and geology contributes to the formation of a unique vegetation type, growth and structure, and therefore it is valid to assume that fuel load quantities and rates of accumulation will also differ.

Little is known of fuel accumulation rates in the vegetation types of the SEQ region. The following paper uses data collected from surveys of Redland Shire Council over the period of 6 months to develop basic fuel accumulation models with which to estimate surface fine fuel loads for two common vegetation types (*Eucalyptus racemosa* open woodland and *Corymbia citriodora/Eucalyptus major* open forest) within the SEQ region.

METHODS

Redland Shire consists of three distinctive regions, the mainland, southern Moreton Bay Islands and North Stradbroke Island. It is bordered by the Pacific Ocean to the east, the city of Brisbane to the north and west and Logan City to the south. The mainland area consists of extant vegetation communities that are mostly open forests or woodlands with occasional closed forests and wetlands (LAMR, 2001). Much like the other areas in southeast Queensland, the Redland Shire experiences a sub-tropical climate, with temperatures ranging from 11-28 degrees Celsius for most of the year (BOM, 2005; RSC, 2005). The mean rainfall for the area is 1322 mm, with the October-April season being wetter than that of May-September (BOM, 2005). Prevailing winds generally come from the south-east (RSC, 2005). Fieldwork was conducted in the study area between March and September 2005. This study was undertaken in the *Eucalyptus major/ Corymbia citriodora* open forests and *Eucalyptus racemosa* open woodlands of mainland Redland Shire.

Sampling sites were selected based on five criteria in an attempt to minimise site-based variations. These individual criteria were; level of disturbance, known fire age, canopy class, topography and soils, and accessibility. A list of observable vascular plants was recorded, along with details of foliage projective cover (FPC). The FPC was calculated using the method outlined in Stock (2005) and Zancola *et al.* (2000) whereby digital photographs were taken of the canopy and converted into black and white pixels by using the Imagepro (v3) computer program (Hacker, 2001). The ratio of black to white pixels indicated the percentage foliar cover at each sampling point. Rainfall data (annual for year post-fire) for each site was obtained from the Bureau of Meteorology (BOM) records for the closest Redland Bay weather station. Five quadrats of 0.5 m^2 at each sampling site were taken. The quadrat was extended 0.5 m above the ground to include near surface fine fuels i.e. small shrubs, grasses and sedges.

The fuel collected was limited to the finer materials (up to 6 mm diameter for dead vegetation and 2 mm for live vegetation) (Tolhurst & Cheney, 1999). The samples were transferred into paper bags for oven drying at 700°C for 72 hours. To avoid contamination and spillage Detpak© brown paper bags were used to

TABLE 1. Vegetational community structure of study sites.

Vegetation Type	**Canopy Associations**	**Understorey**	
Eucalyptus major/ Corymbia citriodora		Grassy	Shrubby
Open Forest	*Eucalyptus fibrosa*	*Cymbopogon refractus*	*Acacia sp.*
	E. microcorys	*Ottochloa gracillima*	*Leptospermum polygalifolium*
Area: 1744 hectares	*E. resinifera*	*Themeda triandra*	*Jacksonia scoparia*
Regional Ecosystem:	*E. seeana*		*Pultanea sp.*
12.11.5	*E. siderophloia*		*Westringia eremicola*
Eucalyptus racemosa	*E. fibrosa*	*Hibbertia stricta*	*Banksia sp.*
Open Woodland	*E. microcorys*	*Cyperus exaltus*	*Hakea florulenta*
	E. seeana	*Fimbristylis sp.*	*Hovea acutifolia*
Area: 1450 hectares	*E. siderophloia*	*Goodenia rotundifolia*	*Melaleuca sieberi*
Regional Ecosystem:	*E. tereticornis*	*Juncus usitatus*	*Pultenaea villosa*
12.9-10.4		*Viola hederacea*	
		Gahnia aspera	
		Dianella caerulea	
		Lomandra longifolia	

(LAMR, 2001)

dry samples. After drying, samples were re-weighed to obtain the net dry weight of the fuel.

Preliminary data analysis indicated the potential for a linear relationship between the measured fuel quantity and the variable/s. Multiple regression analysis was used to examine the relationship between fuel quantity (equivalent t/ha) and time since fire (yrs), foliage projective cover (%), rainfall (1 year post-fire, mm) and fuel depth (mm).

RESULTS

A total of 105 surface fine fuel samples were taken over the period of the study, 5 from each transect on each occasion. Table 2 provides a summary for each site measured, including the vegetation type, the mean fuel depth, the average surface fine fuel quantity (with standard error) the average foliage projective cover (FPC) and the average annual rainfall (1 year post-fire event) at each site. The data is ordered by the time past since the last fire event. The extent of fire history in this study ranged from 0.8-22 years.

For *Eucalyptus major/Corymbia citriodora* open forest total surface fine fuel loads ranged between 6.32-16.09 t/ha with an overall mean (± S.E.) of 12.44±1.34 t/ha. The fuel depth ranged from 8.6-24.95 mm, (mean 18.66±1.78 mm). The deepest fuel bed depth measurement that was taken over the study period was 35 mm with a corresponding fuel age of 11 years. For *Eucalyptus racemosa* open woodland, the total surface fine fuel loads ranged between 3.77-19.51 t/ha with an overall mean of 11.84±1.58 t/ha. The fuel depth ranged from 7.4-34 mm averaging 17.03±2.76 mm. The deepest fuel bed depth measurement taken for this vegetation type was 57 mm with a corresponding fuel age of 6 years.

TABLE 2. Summary of measurements taken at each site in both *Eucalyptus racemosa* open woodland and in *Eucalyptus major/Corymbia citriodora* open forest at Redland Shire, March-September 2005.

Site	*Time Since Fire (yrs)*	*Mean Surface Fine Fuel Load (t/ha)*	*Standard Error of the Mean*	*Mean Litter Depth (mm)*	*Mean Foliage Projective Cover (%)*	*Mean Rainfall 1-year Post-fire (mm)*	*Estimated Surface Fine Fuel Hazard*
Eucalyptus racemosa open woodland							
1	0.8	3.77	0.88	7.4	36.81	98.13	L
2	1.2	4.48	0.59	10.2	30.60	151.33	L
3	2	7.5	1.87	8.8	34.82	60.16	L
4	4.5	15.32	1	19.4	55.06	86.42	M
5	6	8.94	1.76	17.2	Not taken	138.3	M
6	6	6.56	0.22	34	51.7	138.3	H
7	7	11.64	1.55	19.68	14.34	97.86	M
8	8	12.89	0.84	12.70	33.10	77.46	L
9	9	11.90	1.44	12	52.1	102.31	L
10	10	14.35	2.51	Not taken	40.56	76.58	-
11	14	12.54	1.26	17.2	45.88	105.36	M
12	18.5	15.35	2.04	23.8	60.68	119.74	M
13	20	15.68	1.92	17	60.2	98.12	M
14	21	19.51	3.69	18	44.64	110.06	M
15	22	17.19	2.15	21	62.76	151.33	M
Eucalyptus major/Corymbia citriodora open forest							
1	3	6.32	0.48	8.6	31	68.78	L
2	8.5	13.78	1.88	22.40	52.58	121.71	M
3	11	15.70	0.64	24.95	56.35	79.5	M
4	11.5	8.94	1.08	15.60	55.35	104.99	M
5	14.5	13.82	2.10	20	32.71	63	M
6	15	16.09	1.85	20.4	57.16	128.33	M

(L=Low, M=Medium, H=High, VH=Very High fuel hazard ratings estimated from the Overall Fuel Hazard Guide (McCarthy *et al.,* 1999). March-September 2005.

The variation found within the surface fine fuel load clearly indicates the patchy distribution associated with fuel loads across the study area. It also indicates the unevenness of fuel that can accumulate in areas after a fire event. In this study, low fuels were associated with open canopy areas or near fallen logs whereas higher fuel loads were associated in areas with denser canopy cover and noticeable fuel build up near the base of senescing trees or shrubs.

Previous studies (Tolhurst, 1992; Burrows, 2001; Fernandes & Botelho, 2003) have fitted curvilinear relationships between surface fuel quantity and time since fire. This approach is often used as a plateau in the accumulation of fuel load is often reached after a period of time. We employed a similar approach using negative exponential estimation using Statistica© version 7.1 (2005) (Fig. 1). However, there have been other studies (Raison *et al.,* 1983; Fogarty, 1993) where linear relationships have been found for fuel quantity. It was subsequently shown that the linear regression model was more appropriate for our data.

The output from the multiple linear regression model (from SPSS ® v12) is shown in Table 4 and 5. The model suggests that time since fire, fuel depth and FPC are reliable predictors of surface fine fuel quantity and they can independently explain a total of 68% of the variation within the data set. The regression models were subjected to a model fit test based on data residuals.

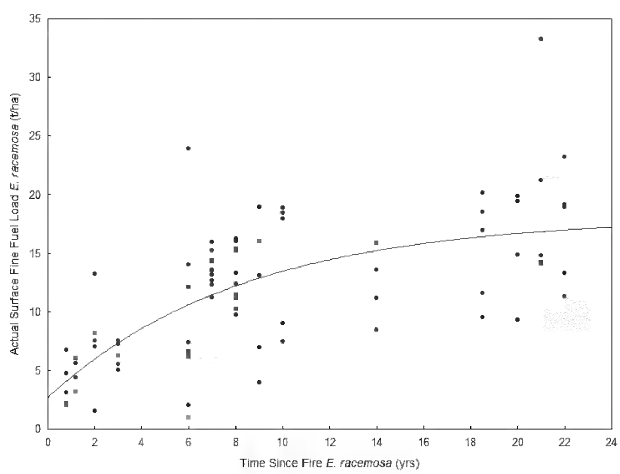

FIG. 1. Curve fitted for the raw data of *E. racemosa* open woodland. Dots represent the raw data. The fitted represents the estimated curve fitted for the data.

TABLE 4. Multiple Linear Regression Model

E. racemosa open woodland	Significance value	Adjusted R^2
Fuel load (t/ha)=x(Time Since Fire)+y(FPC)+z(fuel depth)	0.000	.679

TABLE 5. Linear Regression data output from SPSS for *E. racemosa* open woodland. NB: Bold coefficients indicate statistically significant contribution to regression model.

Model		Unstandardized Coefficients		Standardized Coefficients	t	Sig.
		B	Std. Error	Beta		
2	(Constant)	-.059	1.200		-.049	.961
	Fuel_Depth	**.286**	**.067**	**.355**	**4.293**	**.000**
	Time_Since_Fire	**.321**	**.067**	**.406**	**4.789**	**.000**
	FPC	**.100**	**.027**	**.291**	**3.728**	**.000**

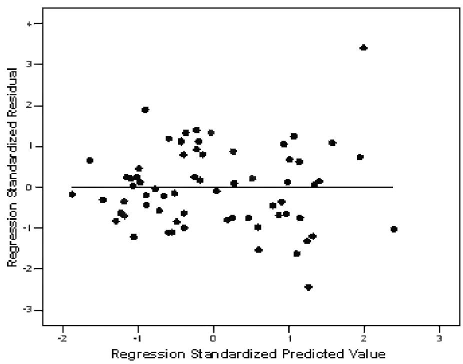

FIG. 2 Scatter plots of the standardised residuals for Model 2 showing the distribution of the data points around the mean for *E. racemosa* open woodland.

Graphing the residual (observed minus predicted) with fire age for each site indicated that the model was not consistently over-predicting or under-predicting fuel weight in the accumulation curve (Fig. 2).

From the analysis it was found that time since fire accounted for most of the variability in surface fine fuel quantity data (Table 5). This was closely followed by fuel depth and FPC. Time since fire and fuel depth explained almost one third of the variation in surface fine fuel samples for *E. racemosa* open woodland.

DISCUSSION

The relationship between surface fine fuel quantity and time since fire, fuel depth, foliage projective cover (FPC) and rainfall (1-year post-fire) were investigated to determine if any or all of these parameters could provide useful and accurate indirect estimation of surface fine fuel quantity. The multiple regression model indicates that fuel depth and foliage projective cover are good predictors of surface fine fuel quantity. Although this study found that rainfall did not contribute significantly to surface fine fuel quantity, other studies have found that rainfall is strongly correlated with litterfall accumulation rates, as high rainfall is generally associated with higher productivity therefore contributing to overall fuel quantity (Simmons & Adams, 1986; Hutson, 1985; Chatto, 1996). The results from this study have provided a basic model that best represents fuel accumulation for two common vegetation types in southeast Queensland.

In line with previous research (Wilson, 1993; McCarthy *et al.*, 1999), this study found a linear relationship with time since fire. It has been noted by several studies (Anderson, 1982; McCarthy, 1996; Sandberg *et al.*, 2001) that fuel depth and structure can directly influence a fire's forward rate of spread and flame height. Chatto (1996) found that 44% of the variation in the surface fine fuel quantities could be attributed to by fuel depth compared to this study where 29% of the variation was explained by fuel depth. Other factors that contribute but were not part of the fuel quantity measurement technique are compaction, sandy substrates, overlying canopy and associated understorey. In this study it was assumed that these influences were to remain relatively constant due to the specificity of the vegetation types. Whilst it is acknowledged that fuel accumulation does not continue linearly with time, the linear regression model provides the best understanding of fuel growth for *E. racemosa* open woodlands in southeast Queensland, for fuel ages up to and including 22 years of age. This is a very useful outcome for land managers to determine appropriate management strategies for hazard reduction or ecological fire management.

There is limited research into the use of foliage projective cover (FPC) as an estimate of surface fine fuel loading in Australia. Brandis & Jacobson (2003) employed the use of satellite images to determine site productivity, providing an estimate of surface fine fuel levels within a given area. This technique provided useful information on fuel loads in the set area, however, it did not directly estimate surface fuel quantity and how this is distributed beneath varying levels of canopy cover which is known to directly affect fire behaviour. This study has determined that FPC has the potential to be used in conjunction with fuel depth and known fire history to provide a reliable estimate of surface fine fuel loads in *E. racemosa* open woodlands. The ability to take a measure of fuel depth along with FPC and accurately estimate the surface fine fuel load will enable a more accurate estimation without the time-consuming and point-specific task of collecting, drying and weighing a number of fuel samples. If successfully tested in other areas it may be able to estimate the time since fire component at those sites where fire history is unknown or subjective. Further study to determine the applicability of this method is recommended. A further step in this project can be to link fuel loads with weather charts or tables to provide hazard ratings for various southeast Queensland forest types.

A technique for the rapid assessment of fuel loads and fire hazards in southeast Queensland eucalypt forests would include the following methodology.

1. Take measurements from within distinct vegetation units – i.e. do not mix measurements taken from different vegetation types.
2. Within each distinct vegetation unit, use a ruler to measure the fuel depth (in mm), from soil surface to upper level of dried surface leaf litter, at no less than 5 separate locations. Collect samples covering the natural range of fuel depths across the site with measurement points at least 20 m apart.
3. At each fuel depth measurement point, also estimate the % foliage projected cover of the canopy above (use % FPC standard charts, e.g. figure 6 in Walker & Hopkins, 1990).
4. Use the average fuel depth figure and the average foliage projected cover in the equation: Fuel load (t/ha) = (0.286 × average fuel depth) + (0.321 × years since last fire) + (0.1 × foliage projective cover).

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FUEL DYNAMICS IN SHRUB DOMINATED LANDSCAPES

M.P. PLUCINSKI, A.M. GILL, AND R.A. BRADSTOCK

Plucinski, M.P., Gill, A.M., and Bradstock, R.A. 2008 06 25: Fuel Dynamics In Shrub Dominated Landscapes. *Proceedings of the Royal Society of Queensland*, 115: 145-151. Brisbane. ISSN 0080-469X.

Changes in wildland fuels with time since fire affect the behaviour and impact of bushfires. The majority of research into fuel dynamics has generally been limited to changes in the load of fine dead surface fuels in forest ecosystems. Fuel dynamics in shrub dominated ecosystems are more complicated as most available fuel is comprised of living vegetation. The historical frequency of fire events influences the composition of plant functional response types and physical structure of post fire regrowth in these vegetation types. The composition and form of post fire vegetation also changes with time due to the lifecycles of the regenerating species. The structural form of the community will subsequently influence the behaviour of a further fire and the conditions under which it will burn.

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Vegetation communities - with biomass dominated by shrubs - include open and closed heath, open and closed scrub, and low shrubland, as listed in Specht's (1970) vegetation classification. Woodlands can also have a significant shrub layer present as an understorey. Many shrublands are renowned for their diversity and have many species that are highly flammable and reliant on fire for regeneration (Gill & Groves, 1981; Specht, 1981; Bradstock *et al.*, 1997; Keith *et al.*, 2001). Shrublands can burn from an early age and support high intensity fires in relatively moderate conditions including a few days after rainfall (Catchpole *et al.*, 1998a; Catchpole, 2001; Fernandes, 2001; Keith *et al.*, 2001; Hammil & Bradstock, 2006).

The bulk of the fuel available to fires in shrub-dominated communities is living vegetation. In other vegetation types, such as forests and woodlands, fires mainly burn in dead fuels such as leaf litter and cured grass and the living canopy often does not burn. The maximum combustible diameter of twigs, branches and stems in shrub fuels varies widely (e.g. Whight & Bradstock, 1999) and is strongly dependant on the weather conditions experienced at the time of fire (e.g. Hammill & Bradstock, 2006). The dynamics of combustible fuel in vegetation communities with significant shrub components are complicated and have received less attention than that of litter layers in forests.

This paper presents a brief review of fuel dynamics in shrub-dominated communities of southeastern mainland Australia. Research investigating pyric succession and fire regimes in these communities has also been considered in order to determine what effects these may have on fuel characteristics in postfire regeneration. The majority of this research has been undertaken in heathlands (vegetation dominated by shrubs less than two metres tall (Specht, 1970)), though much would also be applicable to floristically similar taller shrubland communities and the shrub layers present in some heathy woodlands.

SHRUBLAND FUEL DYNAMICS

Detailed characterisation of fuels is required to understand and predict its effect on shrubland fire behaviour. Characteristics such as the proportion of standing dead fuel, fuel height, and fuel density are important to fire behaviour. Many authors have linked the flammability of older shrublands with high proportions of elevated dead fuels (e.g. Rothermel & Philpot, 1973; Green, 1981; Keeley, 2002), though

some Californian studies have found the ratio of dead to live fuels to be poorly related to stand age (Payson & Cohen, 1990; Moritz *et al.,* 2004). Shrublands with a substantial mass of dead elevated fuel are capable of burning during dry spells even in the coolest and wettest parts of the year (Keith *et al.*, 2001). Vegetation height has been linked with shrubland fire behaviour and subsequently used in some spread rate models (Catchpole *et al.*, 1998a; Fernandes, 2001). Height has mainly been used because it is easily assessed and correlated with other vegetation parameters such as fuel load and density. There have been suggestions in the scientific literature that a variable explaining the spacing of canopy components, such as bulk density, would give a better explanation of the effect of fuel on shrubland fire behaviour than load and height (Fernandes *et al.,* 2000; Fernandes, 2001) based on evidence from laboratory experiments (e.g. Catchpole *et al.*, 1998b). There are relatively few studies that explore the post-fire changes of these fuel dynamics in shrublands.

Fuel accumulation, the net build up of above ground biomass (fuel load) over time, has dominated research into fuel dynamics. The accumulation of shrub fuels is more complicated than litter accumulation in forests and grasslands as there is more influence from the species present at a site and associated growing conditions. There are a number of published fuel accumulation models for shrub-dominated landscapes (e.g. Rothermel & Philpot, 1973; Burrows & McCaw, 1990; Conroy, 1993; Marsden-Smedley & Catchpole, 1995; Fernandes & Rego, 1996; Morrison *et al.*, 1996; McCaw, 1997). These models were produced specifically for quantifying fine fuels consumed in bushfires, usually with a maximum diameter of 6 mm. Most fuel accumulation models are logarithmic equations fitted to discrete data sets for particular shrublands. The upper asymptote, which represents a steady-state fuel load for these models varies from 0.74 kg.m^{-2} (total fuel load) for Banksia low woodland (Burrows & McCaw, 1990) to 4.5 kg.m^{-2} for medium productivity buttongrass moorlands (Marsden-Smedley & Catchpole, 1995). All changes in fuel characteristics have been modelled using time since last fire as the only predictor. However other factors, such as seasonal rainfall variation and browsing or grazing, could potentially influence fuel accumulation and require investigation.

Productivity studies concerned with the total biomass of a stand have considered all fuel fractions, including large woody material not consumed in bushfires (e.g. Specht *et al.*, 1958; Groves, 1965; Groves & Specht, 1965; Specht, 1966; 1981; Jones, 1968a; 1968b; Jones *et al.*, 1969). While these studies have quantified biomass with time since fire, they have not been concerned with modelling fuel development, and site biomass has not always been found to reach a steady-state level.

PYRIC SUCCESSION AND FUEL DYNAMICS

The accumulation of above ground biomass in heathlands is often much more complex than is suggested by models that assume steady-state conditions can be achieved. Floristic and structural changes due to pyric succession can lead to changes in the total fuel load. In heathlands that experience distinct stages of plant regeneration, the relationship between time since last fire and fuel load may not be smooth (see below). Pyric succession involves the gradual ascendance of longer-lived species present in the pre-fire stand rather than replacement of pioneer species with new species. Floristic biodiversity is usually the greatest in the first few years after fire when all of these life forms are present, although there is variation in relationships of observed species richness with time since last fire (Gill *et al.*, 1999).

Specht (1966, 1981) and associated researchers (Specht *et al.*, 1958; Groves, 1965; Groves & Specht, 1965; Jones, 1968a; 1968b; Jones *et al.*, 1969) found a range of relationships between biomass and time since last fire in a number of heathlands in South Australia (Keith, Dark Island) and Victoria (Frankston, Tidal River and Barry's Creek). Many of their results show complex relationships that defy the assumption of a steady-state, and these are reproduced in Figure 1. The saddle shaped curves for heathlands at Keith and Tidal River (Groves & Specht, 1965) and wet heathlands (Specht, 1981) are a result of successional change of plant species in heathlands. In these cases there is a sharp increase in biomass in the first few years that is followed by a plateau before a second increase. The plateau represents a time when the pioneer species associated with the early stages of pyric succession are in decline and the longer-lived shrubs that dominate the long-term fuel load are still developing. This effect can be clearly seen in Figure 2a (Specht *et al.*, 1958), where the delayed return of resprouting species (*Casuarina* spp., *Leptospermum* spp.) and sub shrubs (*Phyllota* spp.) are largely responsible for the second increase in biomass, during the decline of a number of small shrub and herb species (e.g. *Leucopogon* spp.).

The long-lived seed regenerating *Banksia ornata* dominates the community biomass eventually, with the community becoming senescent after 40 or so years. Other studies have also noted the proportion of fuel load from long-lived obligate seeders to increase as vegetative regenerators decline (Ingwersen, 1977; Gill & Groves, 1981; Benson, 1985; McCaw, 1997; Keith *et al.*, 2001).

The trend for a community where a dominant species constitutes the bulk of the biomass from early ages onwards is much less complex, as is the case for mallee broombush shrubland illustrated in Figure 2b (Specht, 1966). Ingwersen (1977) found a similar relationship in a coastal heathland at Jervis Bay NSW.

The decline of a dominant species as it reaches the end of its life cycle can result in a decrease in total biomass and an increase in the proportion of dead aerial fuel. The degradation of senescing shrubs in long unburnt shrublands can open space for the establishment of herb and grass species. The lifespan of a given plant is generally species specific, but may be influenced by site and climate, as well as the presence of competing

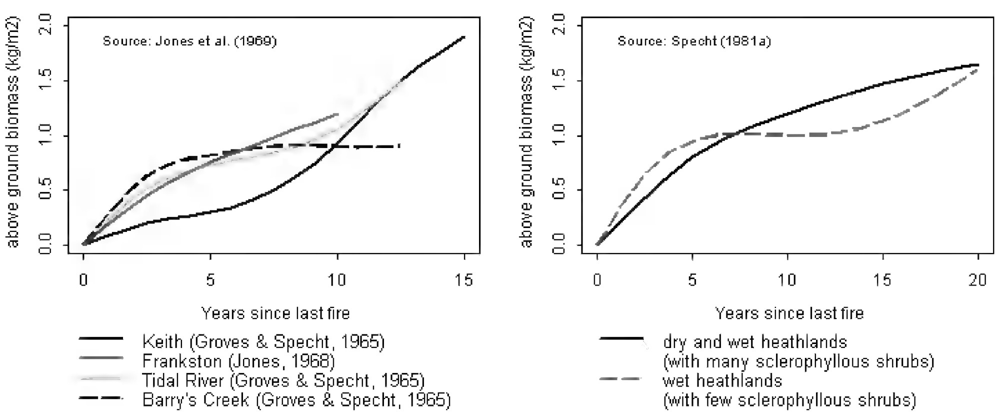

FIG. 1. Heathland biomass accumulation (Jones *et al.*, 1969; Specht, 1981).

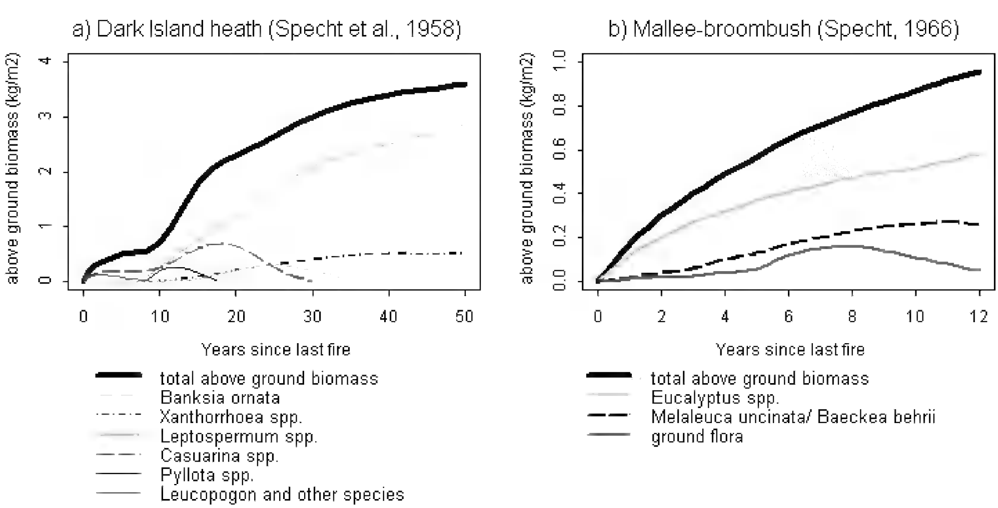

FIG. 2. Biomass accumulation of component species.

plants. The structural and floristic changes from pyric succession can be impacted by disturbances such as grazing (Gill & Groves, 1981). It is likely that other fuel properties, such as percent dead fuel, fuel height and fuel density, are also affected by pyric succession, however their relationships are poorly understood at present.

FIRE REGIMES AND FUEL ACCUMULATION

Fire regimes, consisting of a series of fire events, can affect species composition. Some species, including dominants, can be reduced or even eliminated from the stand as a result of multiple short interval fires. Other species can die out in the absence of fire (Specht *et al.*, 1958; Gill & McMahon, 1986). Both of these contrasting fire regimes may result in strongly altered fuel structures including a much reduced overall fuel load. The state of fuel may therefore be strongly connected to the fate of key species, as determined by the interplay between fire regime components (e.g. length of inter-fire interval) and life-history characteristics. The potential for such effects in heaths and shrublands is high because of the predominance of obligate seeder shrubs in particular and their competitive effects on other species (Keith & Bradstock, 1994; Tozer & Bradstock, 2003). The potential for fire-regime "feedback" into fuel dynamics has been noted for other vegetation types (Bradstock, 1990).

The flora of sites that have experienced frequent fires may be dominated by plants that have short maturation periods or by shrubs that resprout from stems or roots (Specht, 1981). These sites are at risk of losing late maturing obligate seeder species as they are at higher risk of being killed before they can reproduce (e.g. Gill & Groves, 1981; Noble & Slatyer, 1981; Bradstock *et al.*, 1997). If the frequency of fire is very high, repeated fires can also result in the decline of vegetative resprouters (Bradstock & Myerscough, 1988). Repeated fires at short intervals can simplify the structure and composition of heathlands towards an open herb or sedgeland (Ingwersen, 1977; Gill & Groves, 1981; Noble & Slatyer, 1981; Bradstock *et al.*, 1997). In terms of fuel load accumulation, a short interval fire regime may lead to the fuel load increasing at a very rapid rate soon after fire, then reaching a reduced steady-state due to the absence of late maturing large shrubs. In this scenario fuel accumulation would probably be similar to the case of Dark Island heath (Specht *et al.*, 1958) with *Banksia ornata* absent (Fig. 2a). Reduction or elimination of shrubs may lead to a plant community dominated by sedges which, in turn, supports fire behaviour akin to that of grasslands rather than shrublands.

In the absence of fire, the increasing domination of large obligate seeders in long unburnt stands can lead to the development of a tall shrubland (e.g. Specht *et al.*, 1958; Noble & Slatyer, 1981; Specht, 1981; Keith, 1995) (Fig. 2a). The long absence of fire in these landscapes can lead to the loss of the short-lived pioneer species that require fire for reproduction (Bradstock *et al.*, 1997) as well as vegetative resprouters. The seeds of these species are present in the soil after the plants have died, but may not be viable after long periods of time. Obligate seeder species can also be lost if fire is absent for long periods and these species reach senescence (Gill & McMahon, 1986; Keeley, 2002). Obligate seeder species are advantaged by a fire frequency occurring at intervals longer than the primary juvenile period, but shorter than the plant's lifespan (Gill & Groves, 1981). The fuel load of sites that have long fire intervals might experience slower accumulation due to a reduction of short lived pioneer and resprouting species, but may attain a higher load in the long term, as long-lived seed regenerators mature. In this case fuel accumulation would probably be similar to that of *Banksia ornata* in Figure 2a. The peak fuel load would depend on the dominating species and how it grows at a particular site. The type of fires in these taller shrubland fuels would depend on the fuel structure and the openness of the canopy. If the canopy is open, and separated from the ground, surface fires may occur in mild conditions. If the canopy layer is dense and connected to the ground either directly or through ladder fuels, then the whole vegetation stand is more likely to burn and fires might only spread in extreme conditions, as the litter may be unavailable for burning under most conditions.

A choice of target fire intervals to achieve management objectives may require trade-offs that accommodate differing objectives and management values: e.g. the minimisation of potential fire intensity through fuel reduction to achieve property protection versus maintenance of biodiversity (e.g. Morrison *et al.*, 1996; Bradstock *et al.*, 1998; Keeley, 2002). These objectives may be difficult to achieve with a single fire regime (Morrison *et al.*, 1996; Bradstock *et al.*, 1998). Managed fire regimes may aim to vary fire frequency spatially in order to maintain species at a site (e.g. Bradstock *et al.*, 1998; Gill and Bradstock, 2003; Bradstock *et al.*, 2005). An increased awareness of the

influence of species dynamics on fuel dynamics will aid management decision making to achieve multiple values. In particular, where frequent burning may be required to protect adjacent built assets, elimination of dominant shrubs in some areas will alter the fuel array in ways that could make vegetation more ignitable and more prone to faster spreading fires, though the reduction in mass and height of the canopy layer may diminish the maximum level of intensity that could be achieved.

The previous discussion has been summarised in Table 1. The degree of increase and timing of the realisation of a steady state fuel level will depend on individual situations and would have to be verified with field data.

CONCLUSIONS

Fuel dynamics in shrub dominated ecosystems are much more complex than those of forests and grasslands as they include a range of living plant types and are affected by pyric succession. The variety of plant species regenerating in the post fire stand is strongly influenced by fire regime history. These interactions between fire regimes and species composition influence the fuel structure and loading, which in turn impacts on the fuel array and fire behaviour (Fig. 3).

ACKNOWLEDGEMENTS

The authors would like to thank Wendy Anderson and Bob Conroy. This work was supported by an Australian Research Council and New South Wales National Parks and Wildlife Service funded SPIRT grant.

TABLE 1. Hypothetical fuel and fire effects from different fire frequencies in shrublands

Historical frequency	**Low**	**Moderate (managed)**	**High**
Reproductive types	Predominately long lived obligate seeders	All types present in varying proportions	Resprouters, short lived seeders, sedge and grass
Relative fuel accumulation rate	Slow	Moderate/ depending on species	Fast
Steady state fuel load	Very high	High, depending on species	Lower than others
Steady state structure	Tall shrubland	Heathland/ shrubland	Sedge/ grassland
Fire type	Crown fire or surface fire (depending on fuel structure and weather)	Shrub crown fire	'Grass' fire
Fire intensity	Very high (if crown fire), low (if surface fire)	High	Moderate
Fuel availability/ season	extreme weather (crown fire); or moderate weather (surface fire)	High, will burn in moderate weather	Very high, could burn most of the time

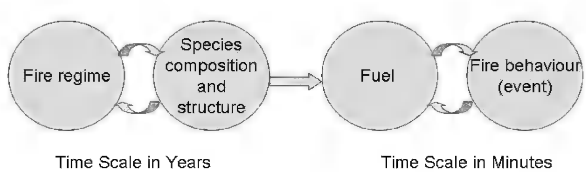

FIG. 3. Flow-on effects of fire regime to fire behaviour

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FIRE, SCIENCE AND SOCIETY AT THE URBAN-RURAL INTERFACE

A.MALCOLM GILL

Gill A.M. 2008 06 25: Fire, Science And Society At The Urban-rural Interface. *Proceedings of the Royal Society of Queensland*, 115: 153-160. Brisbane. ISSN 0080-469X.

The drama of urban-rural interface fire is a feature of summer newscasts in south-eastern Australia. Fire-suppression agencies report on their activities and on threats to homes. At another level, scientists grapple with the problems of predicting fire spread, recommending house-construction methods, advocating human-safety measures and anticipating environmental effects. The householder can be largely unaware of a fire threat or have expectations of total protection from suppression agencies. Houses can burn down and fatalities can occur. This paper considers a number of the issues surrounding this 'bushfire problem'. Using examples based on the fire event experienced under extreme weather in Canberra, Australia, in 2003, simple models and calculations are presented for: the fire-awareness of householders; the proportion of 'knowledgeable' householders; the capacity of the brigade suppression system; demands for water from the mains; stay-or-go recommendations; and, house loss in relation to householder occupancy during fire. A set of testable hypotheses is suggested. The general socio-political problem is how to meet a rare, extreme, short-term demand for resources that far exceeds normal supply. The conclusion that householders need to be self reliant is apparent. The general scientific problem is one of too many variables and too few data for statistical analysis.

❒ WUI, householder responsiveness, water, house occupancy, stay or go, suppression.

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'It is 3.30 on a long hot summer afternoon but it has turned dark from smoke and the street lights have come on automatically. The roar of the approaching fire is intimidating. The gloomy air becomes permeated by a storm of red-hot embers blown into the urban interface by a strong dry wind. People are hosing their houses but gardens are catching alight and many houses seem doomed. There is no fire-suppression appliance to be seen.'

This graphic description depicts the scene of a major bushfire arriving at an urban edge. When houses and lives are lost it is the beginning of a major social problem that can last for years. This is an international problem and one with many facets and complications (Gill, 2005).

Fire, science and society meet at the urban-rural interface and too often in circumstances of death and destruction. What can householders – and society in general - expect? Can science provide any insight into this situation? Can testable hypotheses be formulated, data collected intelligently and better practices put into place?

In this contribution, a limited set of issues is addressed. Here, the house is considered to be the major asset at the interface (see also Gill, 2005). Two 'responsiveness groups' of householders are created and their proportions in the community estimated. Whether or not to stay with the home during a major fire event is considered along with the chances of saving a house using an urban water supply; the situation faced by fire-suppression authorities in such extraordinary circumstances is also considered. This paper is necessarily somewhat speculative but attempts to be constructive and stimulative.

HOUSEHOLDER RESPONSIVENESS TO FIRE

Consider a fire that starts within a hundred metres or so of the urban edge, runs up a slope before a strong dry wind and destroys a house or two (e.g. in the manner of the fire at the edge of the Canberra suburb of Yarralumla, Australian Capital Territory – ACT – in December 2005). In this situation there is very little time for residents to become aware of the threat they face or for agencies to warn anyone of the approach of the fire, let alone respond in time to prevent house loss. The proportion of fire-responding residents in such

cases may follow a curve such as that in Figure 1.

'Responsive' behaviour, here, is considered to occur when a threat has been recognised and action considered, or possibly taken, by the resident in response to it. Action may be limited to mental plans or involve fuel modification, suppression action or a decision to stay and defend the property, or stay and shelter, or move away. The time scale at which awareness is graphed depends on the situation. 'Hours' or 'days' may be appropriate in some situations (see below) as opposed to the 'minutes' of Figure 1: the Canberra, ACT, fires of January 2003 (McLeod, 2003) burned for 10 days before reaching the urban edge of Canberra thereby providing a considerable period for reflection and preparation in response to that event – compared with the general preparedness that is a seasonal routine of some households even in the absence of fire. It was observed by the author from a small sample that most people were completely unprepared in Canberra even close to the time of the fire's arrival, and published narratives or people's experience support this (e.g. see Matthews, 2003).

RESPONSIVENESS CATEGORIES

Two contrasting categories of households are recognised here in relation to their 'responsiveness' to an impending fire. The first consists of those people who are seasonally unprepared, apathetic, unconcerned or vague; they might believe that 'the authorities have everything in hand' or that they are covered by insurance so perceive that they 'need not think about it' or that 'it won't happen to me' or that 'it can't happen here'. This group is called the 'naïve' group. Their counterparts are those who are seasonally prepared, fire-experienced or well informed, watchful, concerned and alert (after Cunningham & Kelly, 1994). This group is called the 'knowledgeable' group. The responsiveness of the two groups in relation to an actual fire starting well away from the interface is speculatively depicted in Figure 2.

According to the hypothetical relationships depicted in Figure 2, all households in which people are present become aware of the fire when it arrives but the proportion of the 'knowledgeable' group aware of the possibility of the impending fire rises many days before that of the 'naïve' group. The 'naïve household' would appear to be more likely to make a decision to stay or leave at the last minute while members of the former category have time to consider their position, make final preparations for the arrival of the fire and their responses to it, and seem more likely to stay and defend their property. It is the impression of the author that in the unprecedented Canberra fires of 2003, the 'naïve' group was the larger of the two but there is no definitive evidence to support or refute this.

Note that according to the hypothetical graph depicting the 'knowledgeable' group in Figure 2 some households are responsive well in advance of any possible fire: this may be seen as being responsive to the chance of ignition, the nature of the fuel array, weather forecasts, and fuel moisture.

Only two categories of responsiveness have been discerned here for convenience. It may be sensible in the future to more rigorously define more groups than the two used for illustration here.

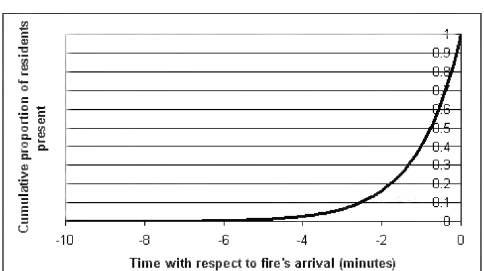

FIG. 1. Hypothetical cumulative proportion of the awareness of fire by a resident urban-edge population. The fire is considered to have started relatively close to the interface.

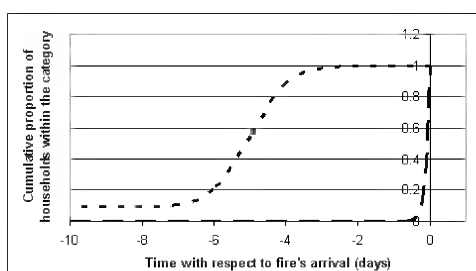

FIG. 2. Hypothetical cumulative proportion of responsive households within the 'naïve' (lower, dashed curve) and 'knowledgeable' (upper, dotted curve) categories in relation to the number days left for the fire to travel before it reaches the interface.

Membership of Response Categories

While responsiveness within a category of responsiveness varies with time, the numbers of households falling into one or the other of the two identified categories varies. Thus, the proportion of 'knowledgeable' households will vary from place to place and at any one place with the passage of time; contributing affects may be the year the resident arrived in the area, the time since the last fire, the effectiveness of official and unofficial warnings, the quality of the information stream from fire authorities, and the extent of personal observation and learning. Residents who have been through a Community Fireguard program in Victoria (Boura, 1999) or are members of a Community Fire Unit (New South Wales and ACT) are more likely – perhaps much more likely – to be in the 'knowledgeable' category than others.

Starting at the end of a socially-disastrous fire, rather than leading up to it as in Figures 1 and 2, we may assume that all affected residents at the interface are part of the 'knowledgeable' group even if they were absent from their homes at the time of the fire because, on return, they would experience the devastation of the neighbourhood, hear the stories of those who stayed and read an extended media coverage of the event. Thus, the major fire event provides us with a starting point for an exploration of year-year changes in the proportion of 'knowledgeable' and 'naïve' households in a community.

In Cunningham & Kelly's (1994) survey in the Blue Mountains, there were 27% "experienced" households (here called 'knowledgeable') 16 years post fire. The model shown in Figure 3 suggests an attrition rate of about 4.67% per year in the proportion of 'knowledgeable' households to 1985 (upper, dashed, line) but then Cunningham & Kelly (1994) identified an additional loss of awareness within the 'knowledgeable' group after 1985 (lower, dotted line); we can call this a 'backsliding' effect. Apathy may follow.

One reason for the decline within the 'knowledgeable' group could be the incorrect perception of residents that the longer it has been since the last fire, the less likely there will be another fire; further perceptions, perhaps misleading, may be that 'these fires only occur cyclically' (Edgell & Brown, 1975); e.g., 'it's not that long since the last one' and 'fire management must have improved since last time' so 'there is nothing to worry about at the moment'.

In suburbs where house numbers are largely static, the sale of houses at the interface after a major fire may be taken to represent the loss of 'knowledgeable' households from the area because 'knowledgeable' residents are more likely to be replaced by 'naïve' ones when a house is sold; new residents would appear to be more likely to be from a city core or from interstate rather than from another interface. Figures for house sales and other housing data in those parts of Canberra directly affected by the 2003 bush fires were obtained from ACTPLA, the Australian Capital Territory Land and Planning Authority. Numbers of sales were able to be expressed as proportions when data for the number of houses per suburb from the 2001 census were obtained from the Australian Bureau of Statistics web site. The ten-year average sales figure – for the period 1996-2005 – was 6.6% per year for the Canberra suburbs of Chapman, Duffy and Holder.

When proportional sales in 2003 were graphed against proportional house losses for the 2003 fires for seven affected suburbs (loss data from the ACT Department of Urban Services), there was a positive correlation (not shown here), supporting the idea of greater turnover of ownership soon after fire. This could be modelled by using a lower value for the 'knowledgeable' group at the outset in Figure 3 rather than assuming a starting value of one, although it is possible that people who sell up in one part of the burnt area then buy up in another.

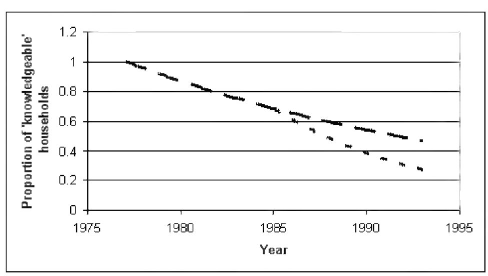

FIG. 3. A model of the proportion of 'knowledgeable' households in the Blue Mountains from the time of the 1977 fire (after Cunningham & Kelly, 1994). The modelled attrition of 'knowledgeable' households, apparently due to a net departure from the survey area of about 4.67% per year, is shown by the dashed line (upper) while the dotted line after 1985 shows the added effect of an apparent loss of 'knowledge' by some households of about 10.9% per year.

The models used above provide us with some indication of what is likely but they cannot be perfect reflections of what actually is the proportion of 'knowledgeable' residents. In particular, the turnover of people in rental accommodation is not known. Residents in such circumstances may be long or short term. The proportion of rented houses in seven affected Canberra suburbs was estimated to be between 8 and 25%. Another complication is the partial sale of houses between joint owners and what effects this might have; partial sales – between multiple owners – were not considered above.

Target audiences for fire-safety messages are likely to be continually changing (Coleman, 1995) as people move into and out of potentially fire-affected suburbs.

EXPECTATIONS OF FIRE BRIGADE PRESENCE: LIMITS TO AGENCY SUPPRESSION CAPACITY

Both 'knowledgeable' and 'naïve' groups may be expecting protection from the urban fire brigade in the event of an urban-interface fire. In 2006 the main categories of people involved in fire suppression in the ACT were the ACT Fire Brigade (urban predominantly), the ACT Rural Fire Service (rural, consisting of Park's units, and volunteers from urban and rural locations, augmented by 'seasonal fire fighters'), the post-2003 Community Fire Units (specially-trained local householders with access to local fire hydrants but under the command of the ACT Fire Brigade), general householders (using mains water from taps - no access to hydrants), and 'farmers' (individual rural lessees). The ACT Fire Brigade and the ACT Rural Fire Service are part of the ACT Emergency Services Agency. The vehicle fleet for house-fire suppression in the ACT Fire Brigade (personal communication, 2006) consisted of 14 pumpers, 5 tankers and, for the first time, 4 compressed-air-foam tankers - a total of 23 possible urban-interface fire-suppression vehicles. Two helicopters were available during the fire season of 2006-07.

On January 18th, 2003, over 500 houses were burnt in Canberra (Leonard & Blanchi, 2005). Assuming one tanker or pumper to each threatened house, then the entire ACT urban capacity is mopped up by just 23 threatened or burning houses at any one time. Given that several thousand houses were threatened and hundreds burnt over a period of several hours in the Canberra fires of 2003 (ACT Government, 2003; Leonard & Blanchi, 2005), the maximum demand for suppression services can greatly exceed supply even if rural fire service vehicles are also used for house-fire protection.

When property protection is paramount, all fire appliances could be fully engaged at rural and urban house sites, so no suppression of the moving fire perimeter by agency fire-fighters is possible. Equipment breakdown and further outbreaks of fire can exacerbate the situation but the timely arrival of crews and appliances from other jurisdictions can offset this. The capacity of rural (Gill, 2005) and urban fire-suppression services can be quickly overcome when many structures are threatened, or already alight.

Would society ever allow governments to spend the money necessary to establish a full-time, fully-equipped, professional fire-suppression force with adequate training for dealing with large, rare, high intensity fires burning into the urban-interface under the worst possible weather and, possibly, fuel conditions? If it did, such supply would be excessive for the vast majority of the time. This problem is at a different time scale to that of normal seasonal variations - which is addressed by having trained volunteers and by employing, apparently increasingly, paid seasonal fire fighters. If an adequate number of volunteers offered their services for the most extreme situation, and they were fully equipped with vehicles, there would be no 'action' for most of them year after year; as a result, many could be expected to lose their enthusiasm and skills and drift away.

Householders can expect that they will be without agency support in the event of a large fire at the urban edge or in the few minutes it takes a fire starting near the edge to reach houses and ignite them; there is an operating domain in which agencies can best assist the public through fire suppression. Defining what this range is in detail remains a challenge. If the time for the fire to reach an urban house from a rural ignition is short, the response time of an urban brigade may be too long to prevent house loss; if the capacity of the fire suppression agency is exceeded because there are too many houses threatened or alight, the response time for many houses may be too long also, albeit for a different reason. Between these limits, however, the response time might be regarded as sufficient.

FIRE SUPPRESSION AND URBAN WATER RETICULATION

The most common resource for all the participants

engaged in fire suppression at the urban interface is water. From the urban perspective, this is supplied by a network of catchment storages, local reservoirs and mains by a government, or semi-government, agency. Farm dams, water tanks and streams supply rural brigades but once near the urban edge, hydrants can be tapped.

Canberra's water is piped from open storages in the mountains to enclosed concrete reservoirs in the hills surrounding the suburbs - from whence it is distributed by gravity to householders through a network of pipes sectored according to reservoir location. A fire may impinge on a long or short edge of a network sector and so affect the demand by householders for water differentially. Neighbours on opposite sides of a street may be in different network sectors and so may experience different water pressures.

While water is the common resource, the tools for its application and the ways in which it is used vary considerably. Urban Brigades have pumpers and tankers of varying capacity and apply water with or without foams (with or without compressed-air enhancement); rural brigades apply water with or without flame retardants (including foams); farmers, householders and Community Fire Units use water without special treatment.

In the urban environment, there can be inadvertent competition for the water from the mains in the event of a major urban interface fire. However, people with independent sources and application techniques – such as swimming pools with pumps and hoses – are free of this.

In the major Canberra fires of 2003, there were two sources of inadequacy in the supply of water for fire fighting and there is a need to distinguish these:

1. The first source was highly localised and apparently characterised by a sudden loss of pressure in the hoses of householders that was not necessarily experienced by neighbours. There were many informal reports of this type. The failure of water pressure in garden hoses was due, perhaps, to the burning through of plastic pipes in garden-sprinkler systems (McFeat, 2004).
2. The second type was experienced as water-supply failure by the urban fire brigade drawing on the mains water supply and this can be regarded as neighbourhood-wide. Demand exceeded supply.

The water pressure at any pipe outlet in the suburbs is influenced positively by the 'head of water' (the difference in elevation between the water level in the reservoir and that at the outlet); water supply is increased by higher pressure and decreased by friction in the pipes and demand on the network. The total frictional losses are dependent on many physical properties of the mains network including the diameter and length of the pipes and, for the householder, the plumbing system of the property. A householder who has joined garden hoses to gain access to a wider area of the property will find a reduced flow due to greater friction.

Having an adequate and alternate water supply to that from the mains – temporary or permanent – is ideal for a householder faced with a major fire event. This may be a swimming pool or a tank, for example. Alternatively, a 'knowledgeable' resident will fill the bath, bins and other containers as a backup in case mains pressure fails. Turning off vulnerable, pressurised, garden-sprinkler systems not in use is advisable.

Modelling pipe network performance under different water-demands, and testing the predictions, is possible for water-supply agencies. If done, weak points in the system could be found and strategies designed for the best use of water in such places.

MITIGATION OF RISK TO LIFE AND PROPERTY

Residents have two basic and obvious options when fire threatens their house: they may stay or they may leave at a time of their choosing. However, there are variations on this theme as some people may be ordered to leave rather than leave voluntarily and some may 'stay and defend' their property or 'stay and shelter' only. Those who leave may do so at various times during the event. Those who stay may try to protect neighbour's houses as well as their own. 'Knowledgeable' residents will, by definition, have a different view of the event than 'naïve' residents, and be better prepared.

LEAVE EARLY

The prevailing paradigm of Australasian fire authorities is one of 'leave the potential fire scene early' if one considers it unwise to stay for various reasons, or 'prepare, stay and defend the home', the rationale being that it is safer to stay with the protection of a building – temporarily, if it catches alight – than to flee from the fire at the last moment (see McLeod, 2003; Handmer & Tibbits, 2005).

'Prepare, stay and defend' or 'leave early' both imply

a necessary minimum time to get ready for appropriate action. In the first instance, time is needed to prepare water sources, don suitable clothing and make last-minute preparations to the house and garden while, in the second, time is needed to exit the potential fire-affected area before the fire arrives and before egress is affected. For the purposes of illustration let us assume that one hour is the absolute minimum time necessary in both cases. Where will the fire be one hour before the fire arrives? If I can see the flames nearby, have I left it too late? Cheney *et al.* (2001) used this approach in the context of fire-fighter safety in a forest fire; here it is applied to the urban-interface dweller.

The maximum predicted distance that a line of fire can travel in one hour on level ground under the extreme weather conditions experienced in Canberra in completely cured grass (i.e. no green grass present) or in eucalypt forest is substantial - 15.5 km, 13.2 and 6.6 km in 'natural', 'cut/grazed' or 'eaten-out' grassland categories respectively (Cheney *et al.,* 1998) and 5.1 km in forest (after Cheney *et al.,* 2001). These are extremes and not the usual of course. However, such long distances would be even longer if there was an upward slope or extreme spot fire activity in the direction of the wind, and somewhat shorter for fire travelling against, or at right angles to, the wind or slope.

Given such rapid rates of spread, leaving for a safe haven when the fire is still far distant is wise if one is going to leave. Note that one hour is not necessarily enough time; one hour is used purely for illustration.

STAY AND DEFEND (OR JUST SHELTER)

Experience from major fires indicates that house occupancy is important to house survival. However, if everyone was at home and capable, there is still a chance that some houses would be lost. On the other hand there is a chance that even if all the homes were unoccupied some would survive. In this section we speculate upon the relationship across the spectrum of possibilities for house loss assuming a general lack of professional fire-fighters. The parameters of the situation could be changed to describe the effects of more or less severe conditions, the presence of more fire fighters, loss of a water supply, house type etc. but there is no attempt to do that here. The idea (Fig. 4) provides a background as to what the situation of 'stay-or-go' might mean in different circumstances.

Actual data are rare and do not cover the spectrum of possibilities. The dashed lines in Figure 4 mark out a domain of house loss; these circumstances are somewhat artificial. As noted above, even if there was 100% occupancy, some houses are likely to be lost under the most severe conditions; even if there was no one present, some houses might survive despite being unprotected. The neat relationships in Figure 4 would be modified as to intercepts and slope, and, possibly, shape, in the real world. The few available data are shown in Figure 5.

The linear extrapolation of the trend line for real data in Figure 5 represents the simplest hypothesis for the relationship between occupancy and house loss at the urban-rural interface.

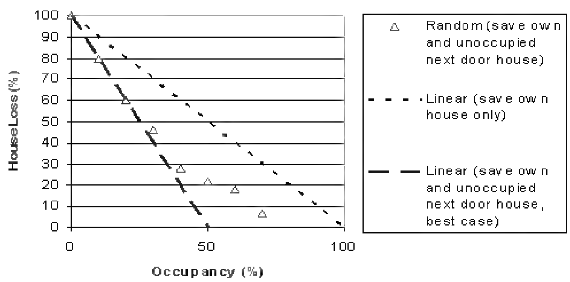

FIG. 4. A model for the loss of houses (n=100) when each household saves its own house and, where indicated, that of a house next door if unoccupied.

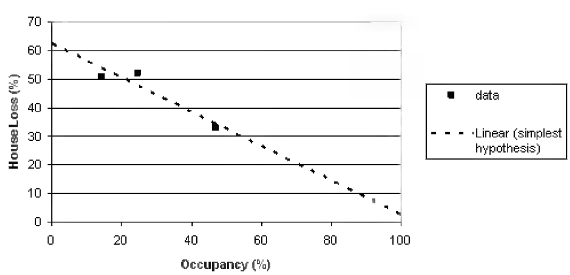

FIG. 5. The dotted line shows the extrapolated trend for data from Wilson and Ferguson (1986) and Leonard and Blanchi (2005, and personal communication for the Otways Fire, Victoria). Note the apparent 62% house loss when no one is present and the apparent 3% loss when everyone is present according to the extrapolation. The real shape of the relationship is unknown.

DISCUSSION AND CONCLUSIONS

There are numerous facets to the problem of landscape fires entering the urban-rural interface (Gill, 2005). Only a small subset of possible topics has been addressed here. Some other topics of importance are: town and landscape planning; building and garden design and construction; disaster recovery; restoration of property and businesses; environmental effects including those on water supply, biodiversity, air and stream quality; warnings; communication of fire information; fire behaviour; and, fuel management. Reports resulting from various official post-fire inquiries canvas many topics which, while not always directed at interface fires, can be relevant there (see, for example, Esplin *et al.,* 2003; Ellis *et al.,* 2004).

'Time scale' emerges as an important variable to consider with respect to fires and the damage they may do at the urban-rural interface during extreme weather conditions. Houses at the urban edge may be affected in a very short time after a fire is ignited nearby; so short can be the time, in fact, that even urban appliances cannot reach the site before damage to property has occurred. On the other hand, the very large fire with long distances to travel may be so large that fire-suppression forces are simply not numerous enough to cope. In both of these cases, the householder who is 'knowledgeable' will be better able to cope. There are limits to effective suppression and a general recognition of these by communities and governments is important if fire problems at the urban-rural interface are to be suitably addressed.

The general problem for society is how to deal with a rare and extreme event like a major unplanned fire burning under extreme weather conditions leading to the loss of homes and human lives – another scale problem. It would appear that the costs of being able to address the most extreme event in a comprehensive way, at least for fire suppression, are prohibitive; the question of the possibility of more comprehensive, but potentially routine, fuel treatments and how they might affect the situation has not been addressed here. How to integrate preparedness and response among the many private and government stakeholders involved when extreme fires reach the urban interface remains an important challenge to all affected, or potentially-affected, societies.

Data issues with respect to the topics of this contribution are substantial. There are too few data and too many variables to consider. This makes conventional scientific analysis impracticable yet policy formulation, even laws, are created or contemplated on the few available data or on perceptions of the circumstances of fires and their impacts on the urban-rural interface. International co-operation in the sharing and analysis of data is recommended. In this contribution, the approach has been to take the scraps of available data and use what appear to be appropriate surrogates, like house-sale information, to develop some testable hypotheses as another step on the way toward greater understanding.

The testable hypotheses arising from this paper follow.

1. Hypotheses related to householder knowledge:
   (a) With respect to their responsiveness to the occurrence of urban-rural interface fires, households can be divided into two groups based on their knowledge, skills and attitude - a 'naive group and a 'knowledgeable' group
   (b) The proportion of households in the 'knowledgeable' category declines predictably in the years after a major fire and is reflected in house-sale data – a surrogate.
2. Hypotheses related to fire suppression:
   (a) Mains water supply at the interface can be accurately modelled. Places less well served than others can be identified.
   (b) Limits to suppression can be accurately modelled. The short response times of urban fire brigades may be demonstrated to be inadequate to save houses under extreme conditions and local ignition while appliance numbers may be too few or inadequate in a large intense fire from a remote ignition point.
3. Hypotheses related to house occupancy:
   (a) 'Occupancy' by able-bodied people is important to house survival. The difficulty here is how to measure 'occupancy'. It could be measured using a score weighted according to 'time-since-fire-arrival' rather than on a simple 'present' or 'absent' basis. For example, based on a negative exponential probability of house loss with time since fire arrival (in the absence of people), the score could be the sum of the probabilities of house loss for each hour, multiplied by a zero or one for the presence of able-bodied people in each time slot. Frequent visits by a neighbour in any time slot would be counted as someone being 'present'.
   (b) House survival is negatively proportional to 'occupancy' by able-bodied people.

(c) The minimum time between safe departure from home and safe arrival at a refuge can be calculated from the fire rate of spread in rural areas and distance of the fire from the house (to give the latest possible departure time for example) and the time to travel to safety during passage from the house to a place of refuge.

There is a need for collaboration between the public, scientists, land managers and governments to learn as much as possible from rare and tragic social circumstances such as devastating fires at the urban-rural interface.

ACKNOWLEDGEMENTS

I would particularly like to thanks the following ACT Government Agencies for their support in this project: ACT Department of Urban Services; ACT Emergency Services Agency through the ACT Fire Brigade and ACT Rural Fire Service; EnvironmentACT; and ACTPLA – the ACT Planning and Land Authority. Scott Byrne of ACTPLA kindly provided data on rentals and house sales and offered interpretive advice. Professor John Handmer kindly commented on the draft manuscript.

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BUSHFIRE BUILDING DAMAGE SURVEY – A NSW PERSPECTIVE

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Douglas G., Midgley S., Tan Z. & Short L. 2008 06 25: Bushfire Building Damage Survey – A NSW Perspective. *Proceedings of the Royal Society of Queensland*, 115: 161-169. Brisbane. ISSN 0080-469X.

Bushfires have damaged or destroyed hundreds of buildings in rural and urban interface areas within NSW for the last decade. These devastating bushfires have taught us many important lessons about bushfire causes and building damage mechanisms. One of these important lessons is that post fire building damage survey is an effective means for studying house damage mechanism and the survey findings can be used to develop better risk assessment models and consequently help us make informed decisions in bushfire risk mitigation planning and development assessment. In recognising the importance of bushfire building damage survey, NSW Rural Fire Service has established a bushfire building damage survey system for collecting, storing, retrieving and researching post bushfire damage survey data. The survey procedure documents when and how to initiate bushfire damage survey. The survey equipment includes laptop computers with GIS functionality, GPS receivers, digital cameras, video cameras and other equipment required for measuring slope and distance. A database has been developed and installed in the laptop computers allowing the survey data to be entered, retrieved and studied. The database captures three kinds of information, that is, general survey information about building and building owner, information required for bushfire attack assessment such as vegetation type, separation distance and slope, and building damage details like degree of damage, building materials of major building components, ignition source and residents behaviour prior to and after fire front pass. The database is not only a survey data storage tool but also an analysing and reporting tool. This paper describes the components of the NSW RFS building damage survey system and the preliminary findings of the survey data collected to date.

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The first scientific bushfire building damage survey was conducted by Barrow G.J. (Barrow, 1945) from CSIR (the progenitor of CSIRO) after a fire at Beaumaris in Victoria in 1944. Since then, most of the bushfire building damages resulting from major destructive bushfire events have been surveyed and researched (Leonard & McArthur, 1999). These destructive bushfire events include bushfires in Hobart (1967), the Blue Mountains (1968), the Otway and Macedon Ranges (1983), Sydney (1994), Sydney (2001) and Canberra (2003). The fintey have also provided useful guidance to assist fire authorities, town planners, architects, developers and practitioners in planning bushfire protection measures as well as prescribing standard construction requirements for buildings in bushfire prone areas (SA, 1999) as well as developing bushfire risk models (Wilson & Ferguson, 1986).

Although the mechanisms of bushfire attack on buildings have been identified and verified by the previous bushfire building damage surveys, it is still necessary to continuously conduct such surveys because of the need of studying the performances of the newly implemented bushfire protection measures and developing or verifying the quantitative models for predicting the probabilities of building loss or survival for a building or a group of buildings under bushfire attack (Leonard & Bowditch, 2003). In recognising the importance of bushfire building damage surveys, NSW Rural Fire Service has established a bushfire building damage survey system for collecting, storing, retrieving and researching post bushfire damage survey data. The survey system comprises a hand-held (Toughbook) computer installed with a survey database which is designed to the standard AFAC/CSIRO (McArthur, 1997) protocol and other data collection equipment such as GPS reader, digital camera and video camera. The main feature of the system is the use

of GPS, GIS and database technologies for collecting, storing, retrieving and analysing building damage or loss data. So far, a total of more than 300 survey records have been collected and stored in the survey database. The majority of these records were obtained through the surveys conducted during the bushfire events of December 2001 to January 2002, October 2002 and December 2002. An analysis of these bushfire damage survey records has been conducted and the preliminary results support the requirements of the *Planning for Bushfire Protection* 2001 (PBP, 2001) and the buffer zone width prescribed for mapping bushfire prone areas in NSW. In addition, the results have also confirmed some existing theoretical or established views in regard to bushfire attack.

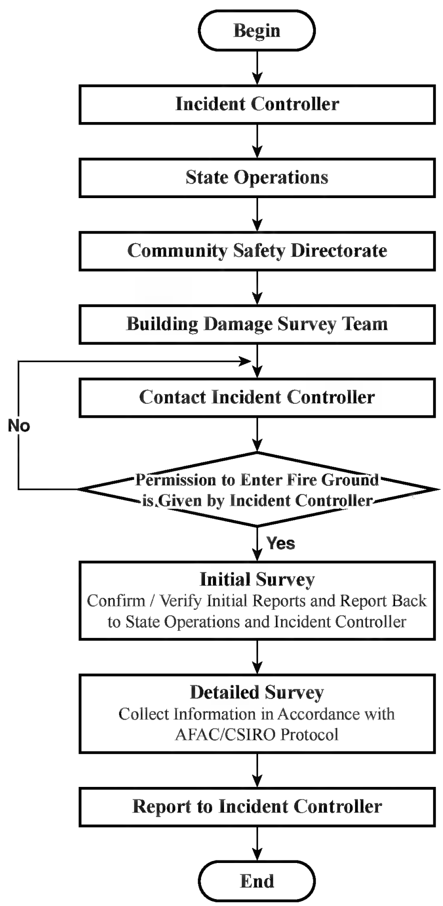

FIG. 1. Bushfire building damage survey procedure

PROCEDURE

The procedure when conducting bushfire building surveys, as shown in Figure 1, is initiated once the survey team within the Community Safety Directorate receives the building damage report from the local IC (Incident Controller) via RFS State Operations. After receiving permission to enter the fire ground from the IC, the survey team will conduct the initial and detailed survey. The initial survey is aimed to confirm and verify the initial building damage report received from the IC while the detailed survey is carried out in accordance with the AFAC/CSIRO building damage survey protocol.

The detailed survey is carried out in accordance with the standard AFAC/CSIRO protocol which is designed to capture three sets of information i.e. the general survey information, the standard site assessment information and the building details. For each set of information, a number of data items need to be collected through the survey (Fig. 2). The major survey equipment includes a hand held computer, which is installed with a database designed to facilitate the on-site data collection and off-site reporting and analysis, a GPS reader, a digital camera and a video camera. The database is designed to the AFAC/CSIRO protocol and linked to a GIS application, MapInfo. The database is able to capture some of the survey information directly from existing GIS data files and therefore accelerates the survey process and improves the survey data quality. In addition, the survey data can be queried and reported in a number of different ways.

DEVELOPMENT OF SURVEY DATABASE

A survey database has been developed by using Microsoft Access. The database is not only able to be used as a central database to store and process all the collected building survey data but also able to be used as a portable database when installed in the laptop computers which allows the survey data to be entered, retrieved and studied in a more efficient and accurate way.

The database is designed to capture three kinds of information: general survey information about building and building owner, information required for bushfire attack assessment such as vegetation type, separation distance and slope, and building damage details like degree of damage, building materials of major building components, ignition source and residents behaviour prior to and after fire front pass. The startup screen of the database shown in Figure 3 provides a graphic user interface which enables surveyors to:

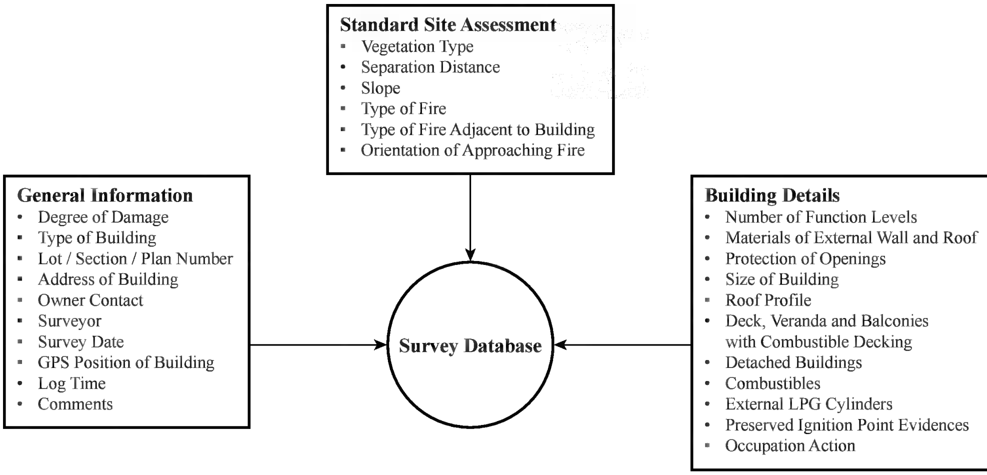

FIG. 2. Information collected during the survey

- Enter/edit the data contained in the interim cover sheet;
- Enter/edit the data contained in Section 1: Standard Site Assessment;
- Enter/edit the data contained in Section 2: Building details;
- Enter/edit the photos taken during the survey;
- Enter/edit the video clips taken during the survey;
- Query the database;
- Preview/print various survey report;
- Exit the application.

Figures 4-6 are three electronic data entry forms which are designed to capture and store the different types of data i.e. general survey information, site assessment information and building details. In order to facilitate users to query the information stored in the database, a user interface for querying has been built within the database.

As shown in Figure 7, the database can be searched in many different ways via the query menu and the query results can be displayed in one of four formats i.e. datasheet, form, text or GIS map as shown in Figures 8-11 .

In addition to the querying capacity, a report menu shown in Figure 12 is built within the database to facilitate the off-site data reporting and analysis process. With the report menu, one can generate a number of different reports. These reports can be previewed or printed in different formats such as text, chart or GIS map.

PRELIMINARY RESULTS

A preliminary analysis of the data stored in the database has been conducted and the results are:

- The cumulative percentage of building ignition varies from the separation distance of building to vegetation (see Fig. 16). The cumulative ignition percentages within 20 m, 40 m, 60 m, 80 m and 100 m are 58%, 76%, 83%, 86%, and 92% respectively. This result is similar to those from the previous research findings (Ahern & Chladil, 1998) and supports both the *Planning for Bushfire Protection 2001* (NSW RFS, 2001) and *AS 3959 - 1999* (SA, 1999) in regard to the adoption of 100 m as the threshold distance beyond which no construction requirements are required.

- The percentages of the ignited buildings which are attacked by the fires approaching from South West and North West are 52.0% and 40.7% respectively being more than 92% together (see Fig. 17). This indicates that the orientation between vegetation hazard and building may have a significant effect on the chance of the building being ignited.

- The percentages of the ignited buildings which take the positions of *Ridge/plateau* and *Top of 1/3 Slope* are 48.3% and 27.0% respectively being more than 75% together (see Fig. 18). This finding further confirms the established view that the buildings sited at such positions are subject to

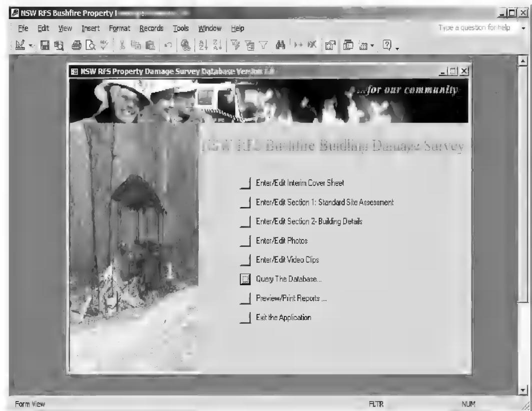

FIG. 3. Start-up screen of the database

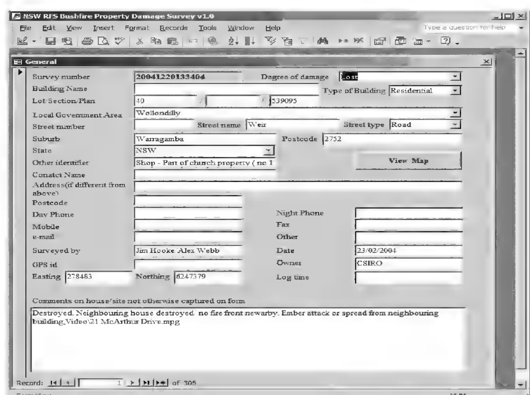

FIG. 4. General survey data entry form

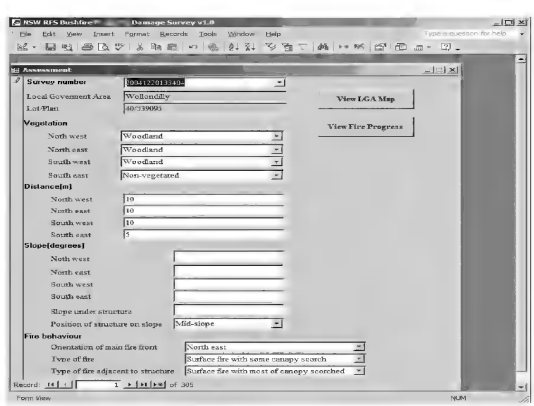

FIG. 5. Standard site assessment data entry form

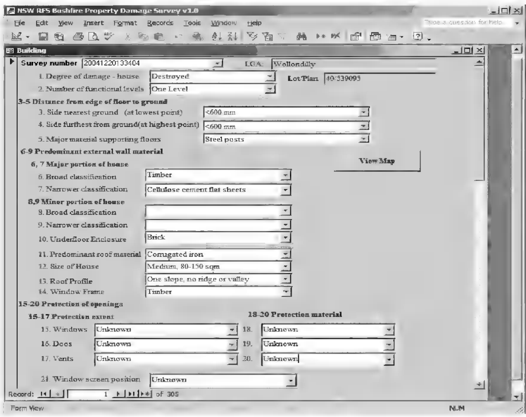

FIG. 6. Building details data entry form

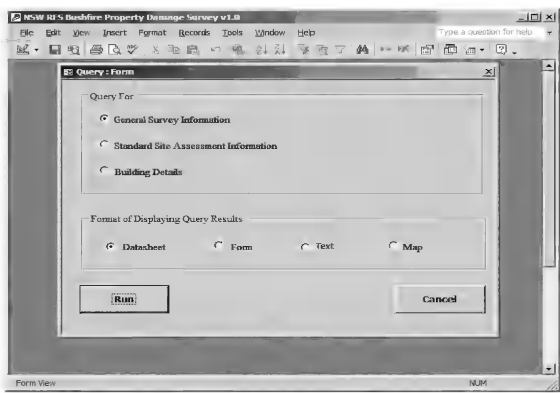

FIG. 7. Query menu

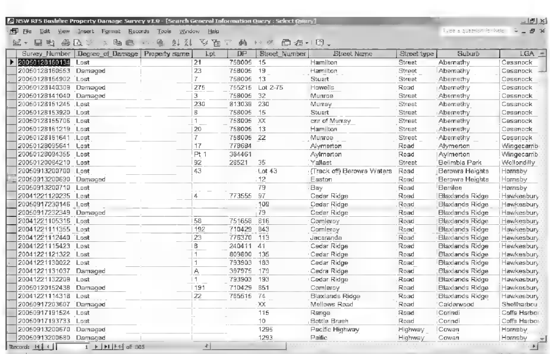

FIG. 8. Sample query results in the format of datasheet

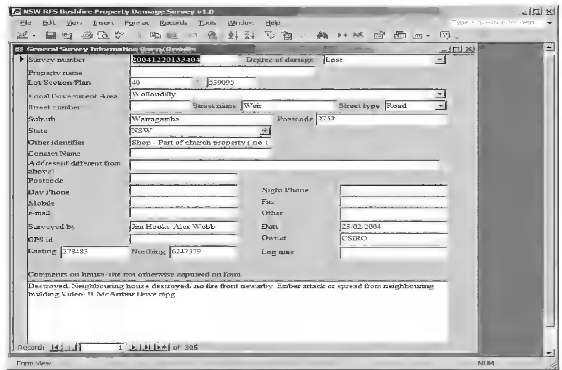

FIG. 9. Sample query results in the format of form

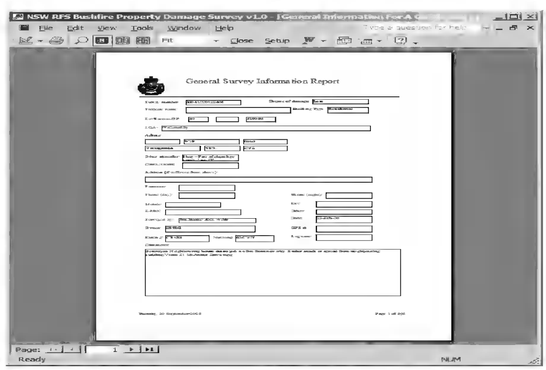

FIG. 10. Sample query results in the format of text

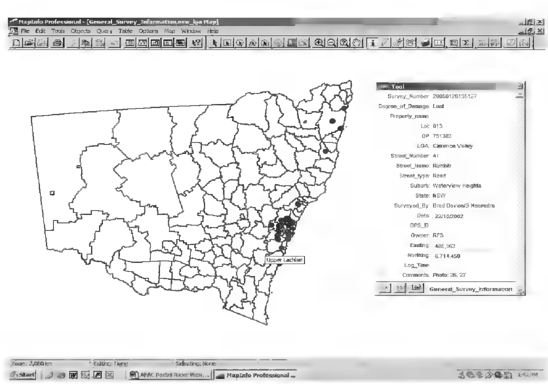

FIG. 11. Sample query results in the format of GIS Map

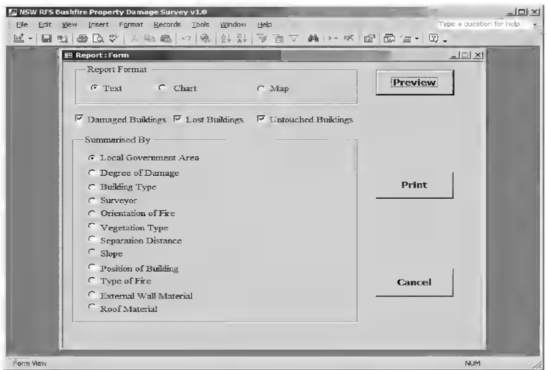

FIG. 12. Report Menu

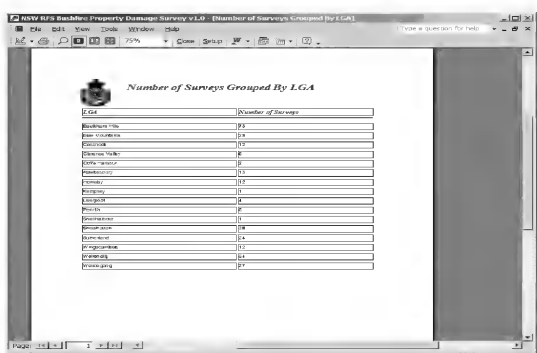

FIG. 13. Sample report in text format

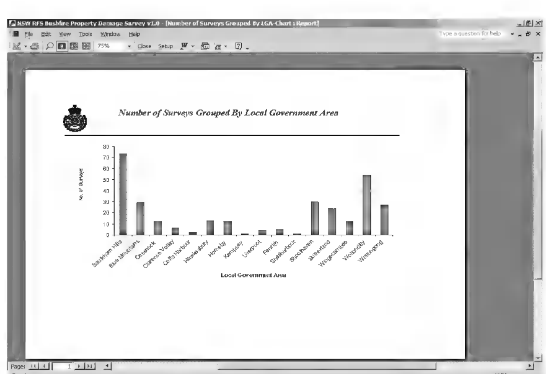

FIG. 14. Sample report in chart format

high level of bushfire attack (Ramsay & Rudolph, 2003).

CONCLUSIONS

- A computer-based bushfire building survey system has been developed and applied in NSW. The system is able to take the advantages of GPS, GIS and database technologies for collecting, storing, reporting and analysing the bushfire building damage information and therefore allows the survey to be conducted more accurately and more efficiently.

- A preliminary analysis of the survey data currently stored in the RFS survey database supports both the *Planning for Bushfire Protection 2001* (NSW RFS 2001) and *AS 3959 – 1999* (SA, 1999) in regard to the adoption of 100 m as the threshold distance beyond which no construction requirements are required and also justify the 100 m buffer zone width for mapping bushfire prone areas for forests and woodlands in NSW.

- The preliminary results confirmed and reinforced the findings from previous building

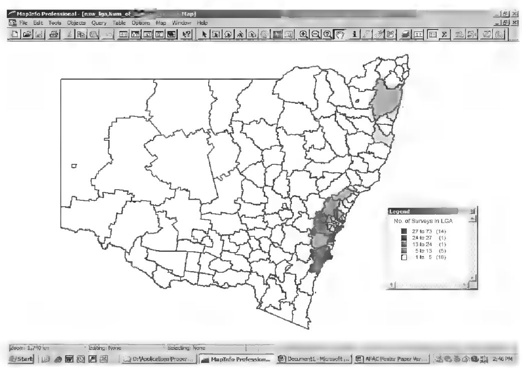

FIG. 15. Sample report in GIS Map format

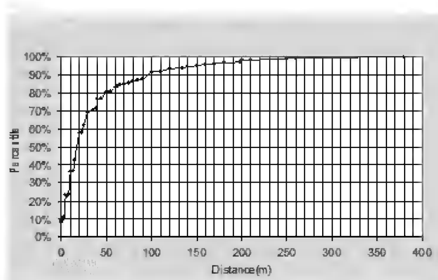

FIG. 16. Correlation between percentage of ignition and distance

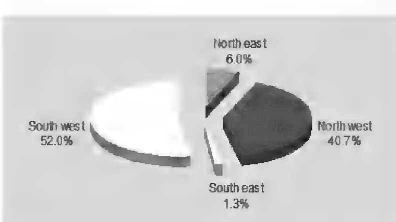

FIG. 17. Number of buildings by orientation of fire

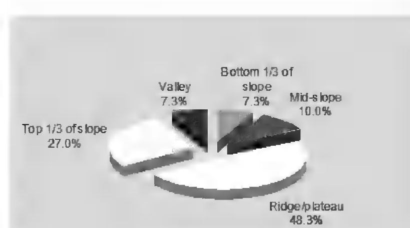

FIG. 18. Number of buildings ignited by position of building

damage surveys (Ahern & Chladil, 1998) and analyses about the effect of separation distance, orientation and building siting on the chance of building being ignited in the event of bushfire.

FUTURE WORKS

- Upgrade the database to further enhance usability, off-site reporting and analysis capacity of the database;
- Develop a suite of SOPs (Standard Operation Procedures) to administer the whole survey process;
- Conduct in-depth analysis of the existing survey data to establish or validate mechanisms of bushfire attack, house loss or survival risk models and effectiveness of the implemented bushfire protection measures.

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AUTHOR PROFILE

Grahame has been involved in bushfire issues in Australia for some 15 years, originally representing community interests on the NSW Bush Fire Council and Bush Fire Coordinating Committee. In 1996, he joined the NSW Rural Fire Service as its first environment officer and has progressed through the organisation in a number of positions within what is now the Community Safety Directorate. He assisted in the development of the *Rural Fires Act 1997*, and has been involved in a number of major changes to the legislation affecting bush fire prevention, hazard complaints, and developments at the bushland urban interface. He was the principal author of Planning for Bushfire Protection (2001) and Planning for Bush Fire Protection (2006). He has also assisted Standards Australia in the revision of Australian Standard AS3959-1999 *Construction in Bushfire Prone Areas*. He is currently guest lecturing in the University of Western Sydney *Graduate Diploma in Design in Bushfire Prone Areas* and teaches in the University of Technology, Sydney *Planning for Bushfire Protection* Short Course.

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